Habitat requirements and conservation of the butterflies *Euchloe ausonides insulanus* (Pieridae) and *Euphydryas editha taylori* (Nymphalidae) in southwestern British Columbia

by
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Abstract

The Garry oak ecosystem of southwestern British Columbia is one of the most endangered ecosystems in Canada. Recovery efforts in this ecosystem are hindered by lack of local knowledge on the ecological requirements of species at risk. Butterflies as a group have declined dramatically in the Garry oak ecosystem. Two species are now believed to be extirpated from British Columbia, the island large marble, *Euchloe ausonides insulanus*, and Taylor's checkerspot, *Euphydryas editha taylori*. The purpose of this study was to synthesize information on the natural history of these species, to document their habitat requirements, and to assess the feasibility of habitat restoration and reintroduction. Observations of *Euchloe ausonides insulanus* suggest that the species could be easily reintroduced to disturbed areas with an abundance of weedy mustard species. Experiments with the weedy mustard host plants show that soil disturbance will be necessary to ensure persistence of the host plant population, and that a large amount of seed will be required to establish a population of host plant. Studies of the habitat requirements of *Euphydryas editha taylori* show that density of host and nectar plants is probably not limiting at potential reintroduction sites, but that host plants senesce too early to support a butterfly population. Mesic areas, where host plant senescence would be delayed, have been eliminated by forest encroachment. Reintroduction of *Euphydryas editha taylori* will not succeed until mesic habitat can be restored through tree removal. Experimental removal of conifers at a historic site of *Euphydryas editha taylori* has resulted in areas that are dominated by exotic plants and do not contain species important to the life cycle of the butterfly. If large-scale tree removal were to proceed in order to

restore butterfly habitat, native plants would need to be actively introduced to treated areas.

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1. Introduction

Garry oak and associated ecosystems are a complex of oak woodlands, meadows, grasslands, mixed forests, vernal pools, and rock outcrops, occurring from south-western British Columbia south to California. These ecosystems occur in areas characterized by mild, wet winters; warm, dry summers; and (historically) frequent low intensity fires, usually set intentionally by First Nations people (Fuchs 2001). In British Columbia, these ecosystems are restricted to southeastern Vancouver Island and the Gulf Islands, with outlying sites in the Fraser Valley and on Savary Island (Figure 1.1).

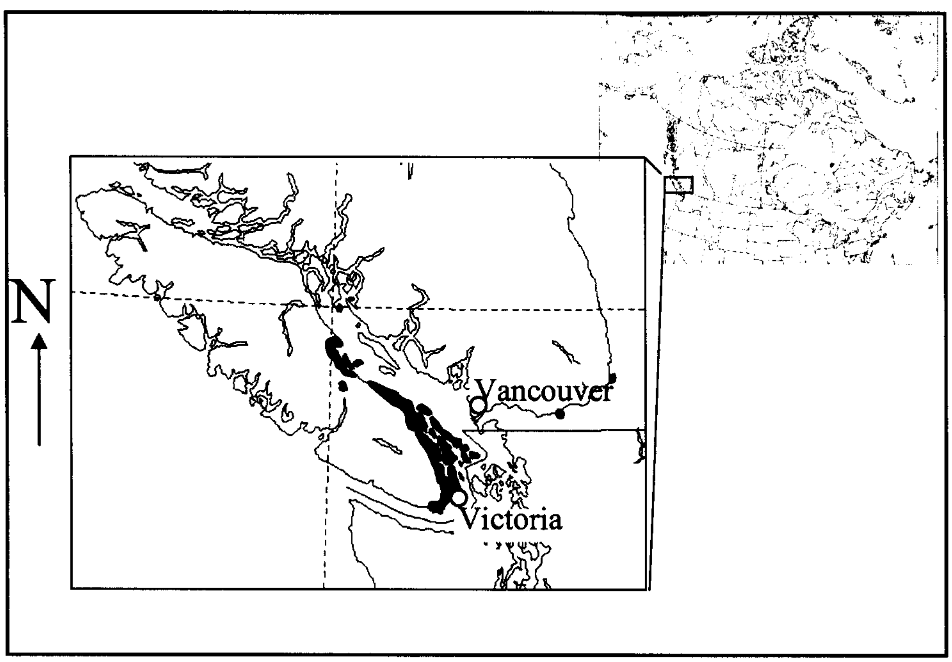

Figure 1.1. The current distribution of Garry oak ecosystems in British Columbia. Modified from Fuchs (2001).

Today less than five percent of British Columbia's original Garry oak ecosystems remain (Lea 2002). Following settlement by Euro-Canadians, these ecosystems were drastically altered and destroyed by the introduction of exotic species, fire suppression, grazing by domestic livestock, and clearing for agricultural and urban development. Remaining fragments are severely isolated, and continue to be threatened by urbanization, exotic species, fire suppression, hydrological changes, abusive recreation, and park developments. The fragmentation of this ecosystem may lead to high rates of extinction, especially as the climate changes.

A great diversity of species are found Garry oak ecosystems, including fourteen herptiles, thirty-three mammals, over one hundred birds, and almost seven hundred plants (Fuchs 2001). While a complete inventory of invertebrates has not been attempted, over eight hundred species of insects and mites have been found to be associated with Garry oak alone (Evans 1985). Many of these species are at risk, including seventy-four plants, two reptiles, three mammals, fourteen birds, one earthworm, and twenty-three insects (Garry Oak Ecosystems Recovery Team 2004).

In 1999, a recovery team was formed to provide direction for protecting, sustaining, and restoring Garry oak ecosystems in Canada. The Garry Oak Ecosystems Recovery Team (GOERT) developed a two-phase recovery strategy (Garry Oak Ecosystems Recovery Team 2002). The objectives of the recovery strategy for the years 2001-2006 (phase 1) are:

1. To develop the information base necessary for ecosystem and species recovery.
2. To protect and manage sites and species-at-risk to minimize immediate losses of ecosystems and species.

3. To motivate public and private protection and stewardship activities by supplying critical information to the appropriate audiences.

GOERT recognizes that these objectives are vague and qualitative. However, the definition of targets in this strategy has been constrained by:

1. Information gaps due to the lack of detailed inventory, mapping, and local research.
2. The large number of species-at-risk and deficiency in information about them.
3. The need for detailed assessment of recovery options as a consequence of extensive habitat loss.

The Garry Oak Ecosystems Recovery Strategy emphasizes the need for more local research on the ecology of species-at-risk, their habitats, and recovery options. Most research on species-at-risk in the Garry oak ecosystem has been conducted in Washington and Oregon (Fuchs 2001). The results of these studies are not necessarily transferable to British Columbia because of differences in ecosystem composition and processes, land tenure, public perception, and environmental legislation.

Butterflies provide an excellent example of changes in the Garry oak ecosystem. Many species were once abundant on southern Vancouver Island. In 1884, Taylor described the extreme abundance of diurnal Lepidoptera as one of the most striking features of the insect fauna of the Victoria area, and stated that almost forty species could be considered abundant (Taylor 1884). In an annotated list of the butterflies of the Victoria area, Danby (1894) described forty species as common. However, by the 1950s, entomologists had begun commenting on the lack of butterflies in the Victoria area (Downes 1956). Despite these declines, however, as recently as the 1960s, hundreds of individuals of many species could still be observed in one day at large fragments of natural land in residential Victoria (J. Tatum per. comm.). Today there is nowhere that this is still true and very few species could be considered as abundant as those described by Taylor.

Twelve species of butterfly that occur in the Garry oak ecosystem have been identified as at risk by the British Columbia Conservation Data Centre (BC Species and Ecosystems Explorer 2003). This ecosystem has been repeatedly identified as an area with a major concentration of threatened butterflies in British Columbia (Guppy et al. 1994, Kondla et al. 2000, Guppy and Shepard 2001). Two of the most endangered butterflies that occur in this ecosystem are *Euphydryas editha taylori* Edwards (Taylor's checkerspot), and *Euchloe ausonides insulanus* Guppy and Shepard (island large marble). Both of these butterflies are endemic to garry oak ecosystems and both are believed to be extirpated from British Columbia (Guppy and Fischer 2001).

The present study is part of a larger project to address critical needs for the conservation and recovery of butterflies endemic to garry oak ecosystems. The overall project has four goals:

1. To confirm or determine the status of butterfly species of conservation concern, including *Plebejus saepiolus insulanus* (island blue), *Euchloe ausonides insulanus* (island large marble), *Euphyes vestris vestris* (dun skipper), and *Euphydryas editha taylori* (Taylor's checkerspot).

2. To describe the life history and habitat needs for selected butterfly species, particularly Taylor's checkerspot and the island large marble.

3. To evaluate the feasibility of re-introduction and potential for habitat enhancement for Taylor's checkerspot and the island large marble.

4. To develop an education and stewardship program for private, First Nations, municipal, regional, and Crown lands directed at the conservation of Garry oak and associated habitats for endangered butterflies, with a focus on Taylor's checkerspot and the island large marble.

The present study focused on goals two and three, and has the following specific objectives:

1. To set the context for butterfly conservation.

2. To document the natural history of *Euchloe ausonides insulanus* and to assess the feasibility of reintroducing this butterfly to its former range in Canada.

3. To document the natural history of *Euphydryas editha taylori* and to identify its habitat requirements.

4. To document changes to the habitat of *Euphydryas editha taylori* in Canada and to begin experimental habitat restoration.

5. To provide recommendations for the conservation of these two subspecies in Canada.

This study contributes to the objectives of the Garry Oak Ecosystems Recovery Strategy by providing locally relevant information on endangered butterflies, assessing possible recovery and conservation options, and raising the profile of endangered butterflies and their habitats. In doing so, this study also lessens the constraints that presently hinder the development of quantitative recovery targets.

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2. **Butterfly conservation**

Insects are the most diverse group of organisms on the planet. The almost one million described species represent approximately eighty percent of the known animal kingdom, and countless millions have yet to be described (Wilson 1987). Insects numerically dominate virtually every terrestrial habitat, in terms of both numbers of species and numbers of individuals. The importance of insects as decomposers, predators, parasitoids, herbivores, and prey items cannot be overstated. Humans rely on insects for many services, including pollination, pest control, soil improvement, and protein (Pyle *et al.* 1981).

Despite the enormous functional significance of insects, their conservation has traditionally received very little attention (Pyle *et al.* 1981, Samways 1993). There are two major impediments to insect conservation. The first is taxonomic (Samways 1993). The sheer number of insect species prevents any researcher or naturalist from being able to identify all but a small sample of the insects in any one area. Many insects can be identified to species only by specialists. Many species remain undescribed, with about seven thousand new species described every year (Samways 1993). It this context, it is very difficult just to determine which species need conservation attention, let alone to formulate a successful conservation plan.

The second impediment to insect conservation is that of public perception (Samways 1993). Despite the fact that less than one percent of known insects behave as pests, the dominant view in modern society is that all insects are pests to be avoided and destroyed. The essential role of insects in natural ecosystems and in the production of human necessities is not widely acknowledged. Very few insects are generally thought of as beautiful, and very few are generally welcomed around the home. In this context, it is very difficult to secure public, political, or financial support for the conservation of those insects whose numbers can be shown to be declining.

One group of insects that has enjoyed unusual conservation attention is the butterflies (New 1997b). Butterflies are uniquely positioned in the insect world as part of a very small group to which the taxonomic and perception impediments do not apply. Butterflies are a relatively small group, and their taxonomy is relatively simple and complete. People without extensive training can quickly learn to identify most species. Many species can be identified even at a distance, and do not require handling or dissection. Butterflies are also one of the very few insect groups that are generally regarded as being beautiful and desirable.

The first instance of conservation concern for a butterfly occurred in the nineteenth century, when a Bavarian state decree ordered protection for *Parnassius apollo* (the apollo) (Pyle 1995). Soon after, British entomologists began expressing concern for the survival of *Lycaena dispar dispar* (the large copper) and *Maculinea arion* (the large blue), while American entomologists expressed concern for *Glaucopsyche xerces* (the xerces blue) and *Oeneis melissa semidea* (the white mountain butterfly). Most of these early warnings blamed the depredations of collectors and the vagaries of weather for declines that were undoubtedly caused by large-scale destruction of habitat (Pyle 1995). The case of *Glaucopsyche xerces* is a notable exception. The decline of this species was attributed to conversion of habitat for residential and agricultural purposes.

Because of the long history of butterfly watching and collecting in Great Britain, the naturalists of that country were well able to detect the declines of butterfly populations. British entomologists thus became pioneers of butterfly conservation, developed many of the techniques that are now in general use, and made crucial mistakes from which the rest of the world could learn.

The early years of butterfly conservation in Britain were dominated by alarm over loss of species with little understanding of the causes of the declines (New 1997b). A major problem was lack of detailed information on what constitutes the necessary features of habitat for a given species, or failure to examine all possible factors affecting a declining species. Because of this, rare species continued to decline even after the protection of apparently good habitat, and reintroduction attempts failed even at seemingly appropriate sites (Warren 1993).

A major breakthrough in butterfly conservation came in the nineteen 1970s, when detailed autecological studies on several species began to reveal cryptic aspects of their ecology and unexpectedly subtle habitat requirements (New 1997b). Once the ecology of rare species was more completely understood, conservation efforts aimed at single species began to enjoy success, and some declining or extirpated populations were restored to a state of health (Pullin 1996). Notwithstanding the increasing success in single species conservation, overall butterfly declines accelerated due to habitat destruction. However, public interest and concern for butterflies began to increase. This growing concern and interest led to the establishment of the British Butterfly Conservation Society (later renamed Butterfly Conservation), which today has a membership of over ten thousand, and engages in activities including monitoring, scientific study, land owner education, and habitat protection.

Arguably, the most important British contribution to the conservation of rare butterflies was the discovery that many species rely on early successional conditions that occur in semi-natural or culturally modified habitats. This discovery followed the observation that many populations continued to decline following the legal protection of their habitats, and that many species declined severely following abandonment of agricultural land or changes to grazing, mowing, or burning regimes (Thomas 1995). This observation led to the initiation of habitat restoration for rare butterflies, as well as providing insights into how to manage land proactively to slow or prevent butterfly declines (Pullin 1996). At many sites where the causes of decline were unclear for many years, resumption of traditional, low-intensity land management techniques has resulted in rebounding and secure populations of rare species.

After Great Britain, the country that has contributed most to butterfly conservation is the United States. The early years of the American experience were similar to those of the British. Alarm was expressed for declining populations, and possible explanations were offered with little scientific backing. Again, a common but unsatisfactory explanation was over-collecting. In the 1930's, a symposium was held on the influence of human civilization on the insects of North America. Participants expressed concern over the potential loss of rare insects, but made no recommendations on how to protect insect populations (Graham *et al.* 1933). By the 1960s, a few detailed ecological studies of rare butterflies were underway, including D. McCorkle's studies of *Speyeria zerene hippolyta* (the Oregon silverspot) (Hammond and McCorkle 1983(84)), and P. Ehrlich's studies of *Euphydryas editha bayensis* (the bay checkerspot) (Ehrlich *et al.* 1975).

A major milestone in American butterfly conservation occurred in the early 1970s, when R. Pyle traveled to Great Britain to study butterfly conservation, and returned to the United States to apply this new knowledge (Pyle 1976). This experience led directly to the creation of the Xerces Society, an organization that practices and promotes global invertebrate conservation, but operates primarily in the United States and deals primarily with butterflies. Since its inception in 1971, the Xerces Society has been a major force in the protection of butterflies and their habitats, and has worked tirelessly to increase the entomological literacy and conservation ethic amongst the public.

In recent decades, autecological studies have become common in American butterfly conservation (e. g. Mattoni 1990(92)), though they have rarely matched the detail of the British studies. In addition, butterfly conservation in the United States has included far less focus on intensive habitat restoration and management and fewer reintroduction attempts. Butterfly conservation in the United States has mainly consisted of charitable societies and government agencies protecting or acquiring habitats of imperilled populations (Pyle 1976). The main contribution from the United States to butterfly conservation may be the demonstration of the utility of effective legislation. The United States' Endangered Species Act has been a powerful tool for the protection of butterflies and their habitats since its passing in 1973 (New 1997b). The listing of a taxon as threatened or endangered under the Endangered Species Act provides penalties for the destruction of individuals or critical habitat, and provides access to funding for the preparation of conservation plans. In addition, the Endangered Species Act provides a legal recourse for the scientific community to have a taxon listed as endangered, even against the will of policy makers. At present, legal action by third parties is the primary method by which endangered species are officially listed.

There is virtually no history of active butterfly conservation in Canada. The Committee on the Status of Endangered Wildlife in Canada (COSEWIC) currently lists sixteen lepidopterans as at risk, although there are at least two major problems with this listing. First, the listing of a taxon presently does not imply or compel any sort of conservation attention. This problem is illustrated by the apparent extirpation of *Euphydryas editha taylori* (Taylor's checkerspot), following its listing as nationally endangered. Despite this listing, the last population was not even being monitored at the time of its extirpation. This problem may soon be assuaged with the implementation of the Species at Risk Act. The second problem is that a great many taxa that are widely considered to be at risk have not yet been assessed by the committee, and there is no indication that they will be soon. This is true even of the many species that have been listed as at risk at a provincial level. This problem is illustrated by the relatively recent extirpation of *Incisalia irus* (the frosted elfin) and *Lycaeides melissa samuelis* (the Karner blue) without their having been assessed by COSEWIC. Conservation efforts on behalf of declining butterflies in Canada have generally represented last minute attempts to save a species that is on the edge of extirpation. Among the Canadian public, it is not widely recognized that butterfly numbers are declining rapidly, or that butterflies are in need of conservation attention.

The science and application of butterfly conservation have developed greatly over the last few decades. A wide variety of techniques have been developed, from simple non-destructive survey methods (Pollard 1977), to complex species-specific monitoring and modeling programs (Murphy and Weiss 1988, Murphy et al. 1990, Weiss and Weiss 1998). Conservation programs aimed at single species have become increasingly detailed, while programs aimed at multiple species have begun to emerge (de Viedma *et al.* 1985, Britton and New 1995, Jelinek 1995). In addition, butterflies have been used as a tool for the conservation and monitoring of other taxa. Butterflies have been found to be sensitive indicators of the naturalness, or conservation value, of some communities (Erhardt 1985, Nelson and Andersen 1994). It must be noted, however, that the effectiveness of butterflies as indicators varies among taxonomic and ecological groups, as well between habitat parameters, so indicator groups must be carefully chosen (Kremen 1992, Kremen 1994). Butterflies have also been found to be useful as umbrella taxa, meaning that their protection may ensure the protection of other rare or declining organisms or ecosystems. Again, however, umbrella taxa must be selected carefully, with reference to the other organisms needing protection, if they are to achieve the desired result (Launer and Murphy 1994, New 1997a).

The most common cause of butterfly decline is habitat destruction. It could be argued that every case of butterfly decline could be traced back to habitat destruction. The solution, then, would appear to be protection of habitat. Certainly, this is usually necessary, but not always adequate. Continued declines of butterfly populations within protected areas have repeatedly been observed (Warren 1993, Prendergast and Eversham 1995, Thomas 1995). These continued declines have driven the science of butterfly conservation biology, necessitating a greater understanding of the habitat requirements of rare species.

Many descriptions of the habitat requirements of rare butterflies begin with the total cover or density of larval host plants, as these parameters clearly have potential to limit populations of phytophagous insects (Dethier 1959, Schultz and Dlugosch 1999, Osborne and Redak 2000). Another parameter that is often measured is the availability and diversity of nectar sources, as this has also been shown to be an important measure of habitat quality for many species (Murphy 1983, Grossmueller and Lederhouse 1987, Hill 1992, Britten and Riley 1994, Loertscher *et al.* 1995, Schultz and Dlugosch 1999). However, defining habitat solely in terms of host and nectar plants does not always lead to success.

Detailed studies have demonstrated that many aspects of habitat suitability are much more subtle than just availability of nectar and host plants. For example, in the case of *Hesperia comma* (the silver-spotted skipper) in Britain, the size of the bunch grass host plant, as well as the state of the surrounding ground cover, are important to the suitability for oviposition. Oviposition occurs only on plants that are about 2 cm in height, 1.7 cm in diameter, and surrounded by about 45% host plant and 40% bare ground (Thomas *et al.* 1986). Plants that are too large, too small, too isolated, or surrounded by too dense a sward, are ignored. In the case of *Maculinea arion* (the large blue) the larvae were known to be brood parasites within the nests of ants of the genus *Myrmica*, but the species continued to decline, even at sites with abundant nests of *Myrmica* species. It was not until the discovery that the butterfly larvae could successfully develop only within the nests of a single species of ant, *Myrmica subuleti*, which itself was in decline because of changing grazing regimes, that the decline of *Maculinea arion* was understood (Thomas *et al.* 1989). In the case of *Euphydryas editha bayensis* (bay checkerspot), long-term studies have revealed multiple unexpected components of habitat suitability. In this subspecies, larval starvation at the time of host plant senescence is the major cause of mortality, and time of host plant senescence is therefore a major determinant of habitat quality. The first major discovery concerning the habitat requirements of *Euphydryas editha bayensis* was that there is a secondary host plant to which larvae may transfer in times of drought in order to complete development (Singer 1972). The second major discovery was that populations occupying topographically complex sites are less susceptible to extinction, as the presence of differing microclimates staggers larval development and, therefore, buffers populations against unusual climatic events (Weiss *et al.* 1988). These studies illustrate the potential problems of concentrating too heavily on the availability of nectar and host plants.

After outright destruction of habitat, the leading cause of butterfly declines is the progressive decline in habitat quality through plant community succession. This is especially important, because succession has the potential to alter habitats and cause declines even in protected areas. Successional changes may lead to excessive shading of host plants when herbaceous communities are replaced by woody vegetation (Thomas 1985, Warren 1985, Sibatani 1990(92), Warren 1991, Marttila *et al.* 1997), or may lead to competitive exclusion of early seral host plants within herbaceous communities (Hammond and McCorkle 1983(84), Singer *et al.* 1993). A second problem that can impair the ability of protected areas to conserve butterflies is that of dispersal and metapopulation dynamics. Conservation programs may fail due to the target species' inability to persist in or colonize sites that are too isolated from other populations (Warren 1991, Thomas *et al.*1992, Thomas and Jones 1993). Declines related to plant community succession can be addressed by habitat restoration or management, while problems related to sites being too small or isolated must be addressed through the acquisition, restoration, or creation of additional acceptable sites. Declines related to isolation may also be addressed through the creation or management of corridors or stepping-stone sites connecting populations (Sutcliffe and Thomas 1996, Schultz 1998, Haddad and Baum 1999).

Many options have been explored in the restoration and management of butterfly habitats. The most direct and labour intensive is the direct planting of plants known to be important to the target species, such as host plants (O'Dwyer and Attiwill 2000). Other techniques include controlled grazing (Pullin 1996), burning (Schultz and Crone 1998), and mechanical removal of woody vegetation (Marttila *et al.* 1997; Warren 1991).

One potential tool in butterfly conservation that has rarely been utilized is captive rearing and reintroduction. That captive butterflies may be successfully maintained over many generations has been demonstrated by breeding programs in butterfly houses, as well as by breeding programs for scientific study (Hughes and Bennett 1991, Lewis and Thomas 2001). Rare species, however, have rarely been bred in captivity (Hughes and Bennett 1991). There is evidence that when butterflies are bred over multiple generations in captivity, they may adapt rapidly to the captive environment, and suffer high mortality when reintroduced (Nicholls and Pullin 2000). Success may be greater when butterflies are reintroduced using wild individuals from a large donor population, instead of from captive populations. Reintroductions using wild butterflies have often failed due to do unsuitability of habitat (Dempster and Hall 1980, Thomas 1989). However, when reintroductions have followed habitat restoration, they have often succeeded (Pullin 1996, Thomas 1989).

The majority of successful butterfly conservation programs share common elements. First, all are based on detailed understanding of the target species and the factors causing its decline. Many authors have commented on the need to base conservation programs on appropriate information, and on the dangers of focussing on partial or inappropriate studies of the biology of the target species (Murphy *et al.* 1986, Murphy 1988, Gaskin 1995). Second, successful conservation programs must include provisions for habitat management, and resources must exist to support this management (Warren 1993, Thomas 1995, Pullin 1996). Finally, many successful conservation programs involve multiple stakeholders, including governments, conservation organizations, private individuals or companies, and the public (Heal 1973, Marttila *et al.* 1997, New 1997b, Sands *et al.* 1997).

While conservation efforts on behalf of single butterfly species have provided a great deal of information on butterfly ecology, the amount of effort required to successfully restore a declining species will not be sustainable as the number of species in decline continues to increase (Ehrlich 1992). There have been few attempts to formulate conservation plans aimed at multiple rare species (de Viedma *et al.* 1985, Britton and New 1995, Jelinek 1995). Increasingly, however, butterflies are receiving attention in regards to management of human-dominated landscapes and in general biotope restoration projects (New 1997b, Smallidge and Leopold 1997). Examples include studies of how agricultural practices influence butterfly diversity in grasslands (Swengel 1996, Dolek and Geyer 1997) and how restoration and reclamation projects can be tailored to increase habitat value to butterflies (Munguira and Thomas 1992, Holl 1996, Ries *et al.* 2001). These projects have the potential to conserve butterflies in a proactive way, in contrast to the traditional approach of planning for conservation only when a species is approaching extinction.

In Canada in general, and on southern Vancouver Island in particular, many species require urgent conservation attention. The impressive studies that have been conducted in Britain and the United States provide insight into how to approach conservation in order to maximize resources and minimize the probability of failure. There are also many species that are known to be declining, though they are not yet critically imperilled. For these species, there is great potential to avoid extirpations and costly conservation programs in the future if critical habitats can be conserved and managed now.

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3. The natural history and conservation of *Euchloe ausonides insulanus*, the island large marble (Lepidoptera: Pieridae)

3.1. Introduction

Euchloe ausonides Lucas, the large marble, is found from central Alaska to California and eastward to the Great Lakes (Opler 1968). Habitats include meadows, grasslands, and forest clearings from sea level to alpine elevations (Opler 1968, Opler 1999). Larval host plants are all in the family Brassicaceae, and include both native and introduced species (Opler 1974).

Between 1861 and 1908, fourteen specimens of *E. ausonides* were collected on Vancouver Island and Gabriola Island in the Georgia Basin of southwestern British Columbia. These specimens represented the only coastal populations of this species recorded north of California (Opler 1968). No records exist of the natural history, host plants, or habitat of these coastal British Columbian populations. *E. ausonides* was believed to be extirpated from the Georgia Basin until 1998, when an extant population was discovered on San Juan Island, Washington (Guppy and Shepard 2001). No other coastal populations in Washington have ever been recorded.

The Georgia Basin populations of *E. ausonides* (Figure 3.1) have long been recognized as distinct from other subspecies, even without formal description (Layberry *et al.* 1998), and have been a conservation priority in British Columbia (Guppy *et al.* 1994). In 2001, these populations were described as a new subspecies, *Euchloe ausonides insulanus* Guppy and Shepard (Guppy and Shepard 2001).

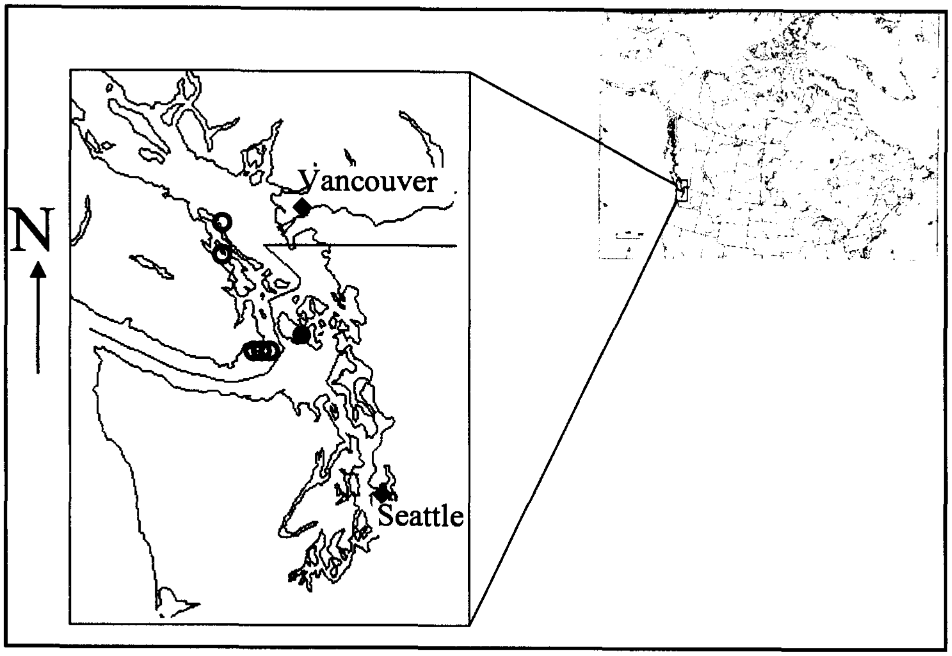

Figure 3.1. The known range of *Euchloe ausonides insulanus*. Open circles indicate extinct populations. Closed circle indicates extant population.

Because all historic sites for this subspecies are in southwestern British Columbia, this is the most logical place to begin considering reintroduction as a global conservation option. The purpose of this chapter is to synthesize personal observations, the observations of others, and information from the literature pertinent to the natural history and conservation of *Euchloe ausonides insulanus*, and to assess the feasibility of reintroducing this butterfly to its historic range in Canada.

3.2. Methods

Because so little was known about this subspecies, the principal approach to this study was direct observation of the butterfly in its habitat, and discussion with other observers. I visited American Camp National Historic Park, San Juan Island, Washington, the site occupied by the only known population of *Euchloe ausonides insulanus*, in May and July 2002, June 2003, and June 2004. During these visits, I observed adults (if present), and searched potential host plants for eggs and larvae. The following findings on the natural history of *E. a. insulanus* are based on observations made during these visits, and on the observations of others who have visited the site (J. Fleckenstein, A. Lambert, and R. M. Pyle). In May 2002, I also participated in a survey of potential habitat throughout the San Juan Islands.

During observations of the habitat on San Juan Island, I noticed that the host plants were growing mostly in areas where the soil had been disturbed by rodents. To determine to what extent the host plants require soil disturbance, I conducted an experiment with two host plants, *Sisymbrium altissimum* L. and *Brassica campestris* L. For this experiment, I examined how the germination of seed of these two species was affected by three site preparation treatments: disturbance of soil by turning with a shovel, mowing of existing vegetation a few centimetres above ground level, and no treatment. There were three plots for each treatment for each species. For each species, three 1 m^2 replicates of each treatment were arrayed randomly in a 3 m x 3 m square (Figure 3.2). Each replicate received one hundred seeds. Seeds were broadcast in November of 2002. In spring of 2003 and 2004, the number of host plants growing in each 1 m^2 plot was recorded. Plants that became established in 2003 were removed prior to seed set, so that no additional seeds would be added. These plots were located on the University of Victoria campus, in an area dominated by tall exotic grasses and weedy forbs, similar to the habitat on San Juan Island. Before treatment, the vegetation of the area was homogeneous, and there were no apparent gradients in moisture regimens or soil characteristics.

Figure 3.2. Plots for host plant germination experiment at the University of Victoria Campus in November 2003, following three treatments.

3.3. **Results**

3.3.1. Habitat

In 2002, I participated in a survey of almost all potential butterfly habitat on San Juan Island, including grasslands, forest openings, rocky hillsides, roadsides, and agricultural land. *Euchloe ausonides* was found only in American Camp National Historic Park, in grasslands and among sand dunes (Figure 3.3). J. Fleckenstein surveyed additional islands in San Juan County, and found no additional populations. In 2003, R. M. Pyle surveyed American Camp National Historic Park, and found *E. ausonides* in marshy areas and among driftwood along sheltered shorelines, as well as in grasslands and sand dunes.

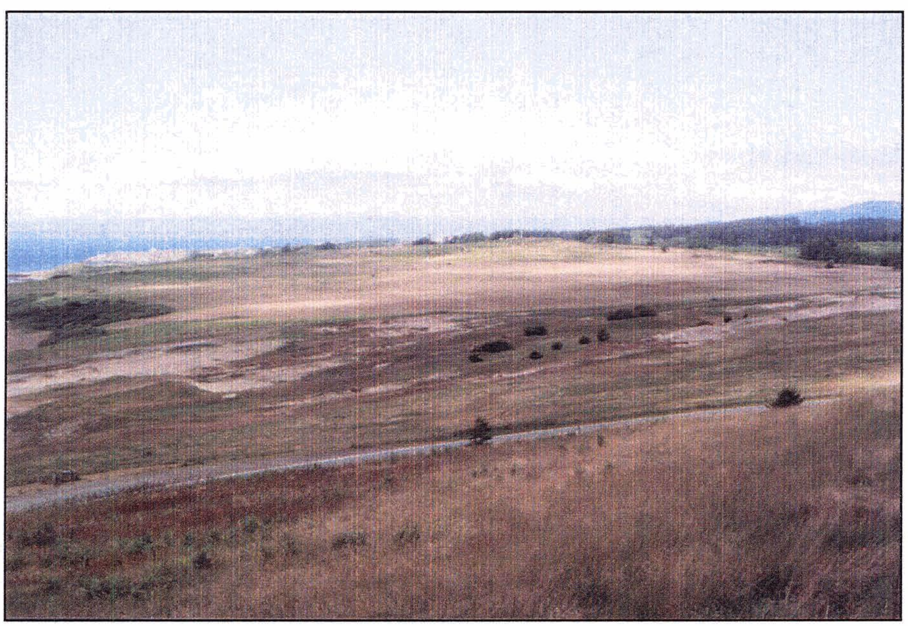

Figure 3.3. The habitat of *Euchloe ausonides insulanus*, showing grassland and sand dunes. American Camp National Historic Park, San Juan Island, Washington.

3.3.2. Life cycle

Adults (Figure 3.4) are in flight from early April to mid-June (A. Lambert, pers. comm.). All eggs that I have found were laid individually on the inflorescence of the host plant. Eggs are bright orange (Figure 3.5). I have not found more than one egg on a single inflorescence, although one plant may receive one egg on each of several inflorescences. Larvae feed on floral parts and developing fruits (Figure 3.6). Most larvae complete development by early to mid-July (A. Lambert, pers. comm.). I have not found pupae. Presumably, larvae leave the host plant to pupate, and overwinter as pupae. If host plant senescence is delayed by favourable weather, a partial second generation of adults may be produced (R. M. Pyle, pers. comm.).

Figure 3.4. *Euchloe ausonides insulanus*, adult.

Figure 3.5. The egg of *Euchloe ausonides insulanus* (orange) on the inflorescence of *Sisymbrium altissimum*.

Figure 3.6. The larva of *Euchloe ausonides* insulanus on *Lepidium virginicum*.

3.3.3. Host plants

During surveys led by J. Fleckenstein in May 2002, eggs were found on *Sisymbrium altissimum* (tall tumble mustard) and *Brassica campestris* (field mustard). I searched these species in July 2002, and found late-instar larvae on both, confirming that they are host plants. Both are introduced species. In 2003, R. M. Pyle found larvae on *Lepidium virginicum* L. (tall pepper-grass), a native species. *S. altissimum* and *B. campestris* are both used in the grassland, where *L. virginicum* does not occur, whereas *L. virginicum* is used along shorelines, where the other hosts do not occur. Only *S. altissimum* is used in the sand dunes, although *L. virginicum* is also present (A. Lambert, pers. comm.). In the sand dunes, *L. virginicum* assumes a low and dense growth form (Figure 3.7.), compared to the upright growth form found along the shorelines (Figure 3.8.).

3.3.4. Nectar sources

The adults that I have observed nectar primarily on the host plants. I have also observed adults taking nectar from *Cerastium arvense* L. (field chickweed) and *Zigadenus venenosus* S. Wats. (meadow death-camas). Both are native species typical of meadows and grasslands.

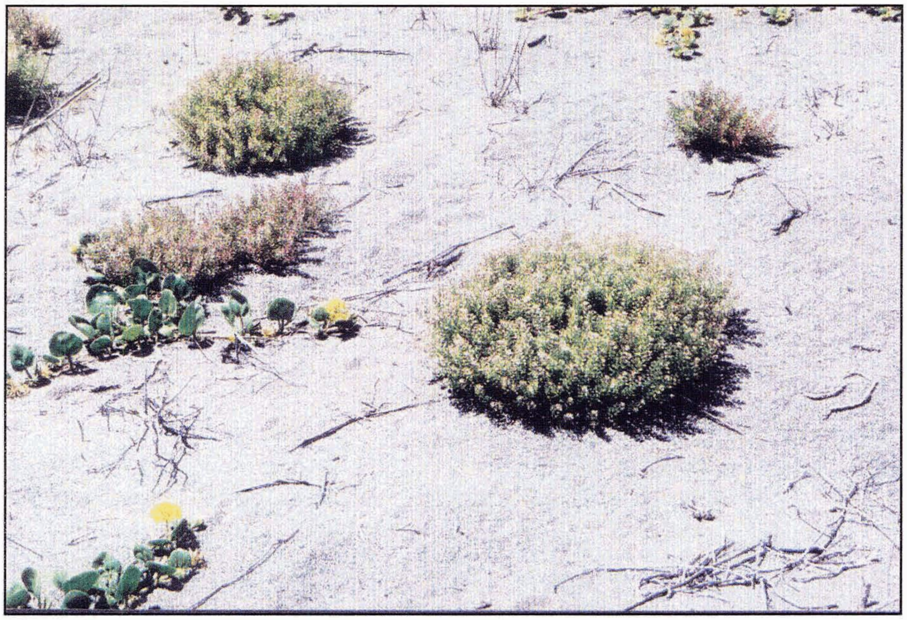

Figure 3.7. The growth form of *Lepidium virginicum* on sand dunes.

Figure 3.8. The growth form of *Lepidium virginicum* along the shoreline.

3.3.5. Sources of mortality

Larvae have been observed being eaten by spiders and birds (A. Lambert, pers. comm.), and are probably eaten by a variety of invertebrate predators. Some larvae are killed by an unidentified disease (Figure 3.9.). No parasitoids have yet been recorded. Small rodents are common in the habitat of the island large marble, and probably feed on pupae from late summer to spring. Birds and invertebrate predators presumably feed on adults.

Figure 3.9. Larva of *Euchloe ausonides insulanus* infected with unidentified disease.

3.3.6. Dispersal

Adults are strong flyers. I have observed individuals flying at least 300m without their landing. However, adults have not been observed outside of the main population at American Camp National Historic Park, except very close to the borders of the park (A. Lambert, pers. comm.). Old-field habitat with an abundance of host plants exists within 5km of the main population, but has not been colonized.

3.3.7. Host plant experiment

Germination rates were very low for both *B. campestris* and *S. altissimum*. In the first spring after seeding, four individuals of *S. altissimum* and three of *B. campestris* became established. In the second spring, an additional three individuals of *S. altissimum* and two of *B. campestris* became established. All plants grew in plots where the soil had been disturbed in November 2002.

3.4. Discussion

Euchloe ausonides has adapted to disturbed habitats throughout its range, and it is not surprising that it has done so in the Georgia Basin. More interesting is the use of the marine foreshore. This is not a usual habitat of this species, and may represent the ancestral habitat of *E. a. insulanus*. The ancestral habitat of this subspecies has previously been assumed to be open grassland areas within Garry oak parkland (Shepard 1995). At present, there is no evidence that this was the case, and the marine foreshore is the only habitat where *E. a. insulanus* is found in association with native host plants. The location tags of the specimens collected between 1861 and 1908 are all vague enough that they could refer to coastal or upland sites. It is possible, however, that before the introduction of weedy non-native species, native mustards, such as *Lepidium virginicum*, may have occurred in disturbed areas of native grassland, supporting populations of *E. a. insulanus*.

Euchloe ausonides insulanus clearly engages in egg-load assessment, meaning that a female's choice of whether or not to oviposit on a particular plant is influenced by the presence of conspecific eggs on the same plant (Shapiro 1981). This behaviour is common in pierid butterflies (Shapiro 1981). Because females recognize conspecific eggs, no inflorescence receives more than one egg. Egg-load assessment has been found to increase larval survivorship by reducing intraspecific competition and incidence of cannibalism (Rausher 1979).

The possibility of a partial second generation in *E. a. insulanus* is unusual. In most species of *Euchloe*, the pupa overwinters in an obligatory diapause (Opler 1974). In California, however, *E. ausonides* responded to the introduction of a summer-flowering host plant, *Brassica nigra* (black mustard), by producing a partial second generation (Opler 1974). This suggests that pupal diapause in this species is facultative. It may also point to a possible Californian origin of *E. a. insulanus*. A Californian origin is likely, considering the habitat of *E. a. insulanus*, and the biogeography of the Georgia Basin. Many species typical of California spread to the Georgia Basin during a warming period between four and seven thousand years ago (Hebda 1993).

The use of introduced mustards by *E. ausonides* is widespread (Opler 1974, Shapiro 1984). Many mustard-feeding pierid butterflies readily oviposit on novel host plants (Shapiro 1975, Chew 1977). However, in some cases, females will oviposit on plants that lead to high mortality of the resulting offspring, and in some cases, larvae will feed and develop normally on plants that are ignored by ovipositing females (Bowden 1971). In the case of *E. a. insulanus*, late-instar larvae have been found on all known oviposition plants, confirming that they are also acceptable larval host plants. Whether growth and mortality rates differ between host plants is unknown. However, larvae appear to feed more readily on the native *Lepidium virginicum* than on the introduced host plants. On the introduced hosts, especially *Sisymbrium altissimum*, larvae feed only for short periods of time and spend a lot of time wandering, while on *L. virginicum* they spend much more time feeding, and do not hesitate to feed (A. Lambert and R. M. Pyle, pers. comm.). The larvae of related butterflies exhibit similar hesitant and restless behaviour when feeding on plants with tougher siliques (Wiklund and Ahrberg 1978).

It is interesting that *Lepidium virginicum* is an oviposition plant on the foreshore, but not in the sand dunes. In the sand dunes, *L. virginicum* grows as low and compact bunches, with many short inflorescences (Figure 3.7.). In California, Shapiro (1981) found that *E. ausonides* rarely oviposits on plants less than one metre tall. In a later paper, Shapiro (1984) concluded that phenophase of the oviposition plant is probably more important than height (or species), in that oviposition occurs only on plants that are bolting into flower. It may be that ovipositing females do not recognize *L. virginicum* in the sand dunes because of the lack of a well defined inflorescence rising above the rest of the plant, or that the plants are simply too low-growing to be recognized. Supporting this idea is the fact that two other mustards are common in the grasslands, but do not receive eggs (A. Lambert and R. M. Pyle, pers. comm.). These species, *Cardamine oligosperma* Nutt. and *Teesdalia nudicaulis* L., are both low growing, although they do have well defined inflorescences. All oviposition plants at American Camp National Historic Park have an upright growth form and clear inflorescences.

Despite the strong flight of individuals, *E. a. insulanus* does not appear to be a strong disperser. No individuals have been recorded on other areas of San Juan Island outside of the main population, not even in relatively close (within 5 km) agricultural fields with abundant host plants. This is surprising, as *E. ausonides* in other areas has been found to sometimes fly distances of at least a few kilometres (Scott 1975). It may be that *E. a. insulanus* has evolved a more sedentary habit as an adaptation to relatively harsh and windy coastal environments. Individual flights are, however, long enough to conclude that the butterflies at American Camp National Historic Park do represent a single population, and not a metapopulation. Individual butterflies disperse across the landscape, and move readily between the grassland, the sand dunes, and the foreshore. Related species have been found to exchange individuals across distances of at least several hundred metres, even when physical barriers are present (Emmel 1972 (1973)).

These observations of the single extant population of *Euchloe ausonides insulanus* fail to provide insight into the extirpation of this butterfly from British Columbia, probably prior to 1910. It has been suggested that, because of the small number of specimens collected, this species must always have been rare in the Georgia Basin (Shepard 2000). However, one author described the species as common in one area of Victoria, British Columbia (Danby 1894). Probably the species was restricted to very few sites, where it may have been fairly common, as is the case with the extant population. The most plausible suggestion is that historically intense sheep grazing may have eliminated the few British Columbian populations of this butterfly (Shepard 1995, Guppy and Fischer 2001, Guppy and Shepard 2001). This is consistent with what is known of the life history of the butterfly, as the eggs and larvae would be very sensitive to grazing, mowing, or haying, given their position high in the inflorescence. However, the grasslands at American Camp National Historic Park were also heavily grazed by sheep, and the foreshore, possibly the ancestral habitat in British Columbia, probably was not heavily grazed. Possibly the ancestral habitat was actually open grassland areas with native mustards, which would have been heavily impacted by grazing. The foreshore habitat at American Camp National Historic Park may have been a unique habitat that served as a refuge during heavy grazing, thus saving the subspecies from extinction.

Although the single extant population of *Euchloe ausonides insulanus* is apparently thriving, and exists within a protected area, the future of any butterfly that has been reduced to one population must be considered precarious. The security of this subspecies would be greatly improved by establishing additional populations through reintroduction. Because *E. a. insulanus* has never been recorded from elsewhere in Washington, the most logical place to begin reintroductions is the subspecies' historic range in Canada. Because of the weedy nature of the larval host plants, and the apparent lack of additional important habitat features (i.e. nectar sources, structure), it should be possible to create new habitat by seeding the host plants into areas with low conservation value, such as old agricultural fields or roadsides. Results of the host plant germination experiment suggest that if this is to be done, periodic soil disturbance will be necessary to maintain host plant populations, and that a large amount of seed will be required initially. Low germination rates and multi-season dormancy are common in plants of the

Brassicaceae family (Hillhorst 1997, Landbo and Jorgensen 1997, Boutsalis and Powels 1998, Mehra et al. 2003).

If new habitat is to be created, it should contain as many species of acceptable host plant as possible. It is likely that *E. a. insulanus* would utilize a variety of other introduced and native host plants, provided they were of appropriate size and phenophase during the flight season. In species closely related to *Euchloe ausonides* polyphagy increases population persistence, as the host that confers the highest larval survival varies according to climatic conditions (Chew 1975, Wiklund and Ahrberg 1978). The best host plant in a wet year may senesce too early during a dry year, leading to larval starvation. Similarly, the best host during a dry year may be in too cool of a microsite during a wet year, retarding larval growth. This situation is analogous to those of other butterflies, where population persistence increases with increasing topographic, and therefore microclimatic, heterogeneity (Weiss et al. 1988).

This study has shown that, while the cause of the historic decline of *Euchloe ausonides insulanus* will never be known with certainty, the butterfly apparently has fairly simple habitat requirements. It should be possible to create acceptable habitat for this animal within its historic range in Canada, and begin reintroductions.

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4. The natural history and habitat requirements of *Euphydryas editha taylori*, Taylor's checkerspot (Lepidoptera: Nymphalidae)

4. 1. Introduction

Euphydryas editha Boisduval (Edith's checkerspot) is found throughout western North America, in habitats ranging from coastal chaparral and grasslands to montane forest openings and alpine tundra (Opler 1999). Throughout its range, this species occurs as discreet populations that are highly adapted to local conditions. Host plants are mainly in the families Plantaginaceae and Scrophulariaceae. Many host plants are used by the species, although each population is restricted to very few (White and Singer 1974). *Euphydryas editha* has been the subject of intensive ecological studies for more than forty years, and may be the best-studied butterfly in the world. This species has declined in many parts of its range. Subspecies *bayensis* and *quino* are listed as endangered in the United States, and subspecies *monoensis* and *taylori* are candidates for listing as endangered.

In the late nineteenth century, larvae of *Euphydryas editha* were collected from Beacon Hill Park in Victoria, British Columbia, by G. W. Taylor, and reared to adults by W. H. Edwards. These adults were originally described as a new species (Edwards 1888), and later recognized as a subspecies of *Euphydryas editha*. At the time of its discovery, *Euphydryas editha taylori* Edwards was considered one of the most common butterflies of the Victoria area (Danby 1894). At least fourteen populations existed in greater Victoria (Hardy 1956, Shepard 2000, J. Tatum pers. comm.). An additional six populations were recorded north of Victoria on Vancouver Island and Hornby Island (Shepard 2000). In the United States, at least forty-seven populations were recorded from the rain shadow of the Olympic Mountains and the south Puget Sound in Washington State, and the Willamette Valley in Oregon (Figure 4.1) (Vaughn and Black 2002).

Populations of *E. e. taylori* have declined dramatically throughout its range. Currently, two populations are known in Oregon and twelve in Washington. Most of these populations are small, and very few are on land that is managed for the purpose of biological conservation. In British Columbia, only two large populations were known by the 1980s. By the 1990s, only one of these remained (Guppy and Shepard 2001). By 2001, this population was gone, and *E. e. taylori* was, therefore, extirpated from British Columbia and Canada.

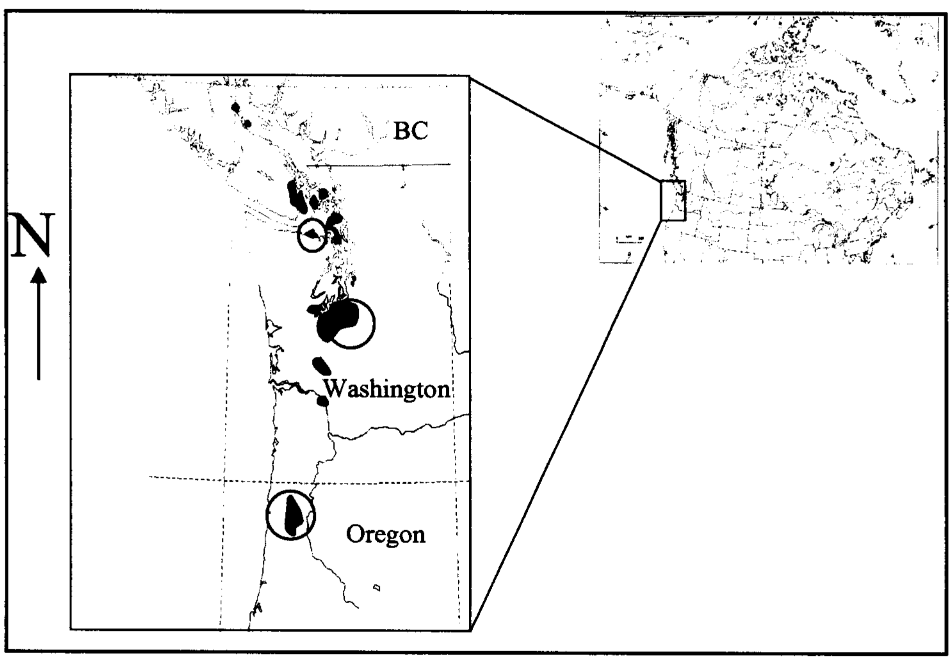

Figure 4.1. The historic distribution of *Euphydryas editha taylori* (shaded). Circles indicate areas where extant populations are located.

Many populations of *Euphydryas editha taylori* have been lost because of habitat destruction for agriculture or residential developments. However, populations have also been lost from protected areas. Before their extinction, some of these populations were considered healthy, and the habitats they occupied were considered to be of high quality (Shepard 2000). It is therefore necessary to describe the habitat requirements of *E. e. taylori*, in order to better define what constitutes good habitat and so provide a scientific base for conservation. The purpose of this study is to describe the natural history of *Euphydryas editha taylori* and to identify its habitat requirements.

4. 2. <u>Methods</u>

Basic ecological information on *Euphydryas editha taylori* is incomplete and poorly recorded. Therefore, a major objective of this chapter is to synthesize available published information and the observations of those who have worked on this animal in recent years. As well, I visited four extant populations of *E. e. taylori* in Washington and Oregon during the flight season in May 2002, April 2003, and May 2004. During these visits, I observed adults, larvae (if present), and eggs (if present), and consulted with other researchers and conservation authorities. These visits and discussions, together with information from the literature, form the basis of my account of the natural history of *E. e. taylori*. I also visited many British Columbian sites from which *E. e. taylori* has been extirpated. These visits allowed me to consider the range of habitats formerly occupied by the butterfly, as well as the range of potential threats.

I described features of habitats at three sites in the Victoria area and three sites on Hornby Island in June 2002. In each location, I chose two sites that had once supported a population of *E. e. taylori* and one site that had not, but was considered to have potential as a reintroduction site. At each site, I determined the density of host plants and nectar sources in 0.5 m^2 circular plots placed regularly throughout the site. This is similar to methods used by Dobkin *et al.* (1987) to determine host plant density. Distance between plots varied between sites in order to assure a reasonable number of plots at small sites, as well as a manageable number of plots at large sites. Density of host and nectar plants between sites was compared using ANOVA. Arcsine transformation of data to improve normality did not affect ANOVA results, and so is not presented. Significant differences were determined using Scheffe's Test.

To describe the habitat as experienced by a feeding larva, I located the closest host plant to each of the plots described above and recorded the ground cover in a 1 m^2 circular plot centred on the host plant. This plot size was chosen based on the finding that larval habitat selection is not affected by the structure of vegetation more than 0.5 m away (Osborne and Redak 2000). Groundcover was categorized as host plant, shrub, herbaceous cover < 30 cm tall, herbaceous cover > 30 cm tall, rock, or unvegetated (including bare soil, bryophyte, and thatch). These categories were meant to encompass features of microhabitat known to be of importance to a feeding larva, such as proximity to other host plants, levels of shading, and proximity to areas for basking. Percent cover for each of these categories was compared between sites using ANOVA. Arcsine transformation of data to improve normality did not affect ANOVA results, and so is not presented. Significant differences were determined using Scheffe's Test. All statistics were calculated using the program SPSS 12.

To investigate the importance of host plant phenology, I marked individuals of *Plantago lanceolata* L. at 10m intervals in a 50m x 70m area in Helliwell Provincial Park, site of the most recently extant population of *E. e. taylori* in Canada. The area was chosen because it was within the area formerly occupied by *E. e. taylori*, and it was the largest possible area that did not contain trees, maximizing the number of plants that could be located at regular intervals. I visited these plants every two to three weeks throughout the spring and summer of 2003 and recorded their state of senescence. Senescence was recorded on a scale of one to four, one being green and turgid, two being green and wilted, three being partially brown, and four being entirely brown.

4. 3. **Results**

4.3.1. Habitat

Euphydryas editha taylori presently occurs in grasslands, estuarine meadows, dry hillsides, openings in Garry oak woodland, and anthropogenic forest clearings. All sites of extant populations are small areas, and most are bounded entirely by forest. Habitats range from areas dominated by exotic plants to areas showing minimal exotic plant invasion and a high diversity of native species. Sites generally have very low coverage of trees and shrubs. Sites of extant populations range from sea level to approximately 600 m in elevation. Sites of extinct populations are mostly dry hillside and grassland, but also include maritime meadows and anthropogenic forest clearings.

4.3.2. Life cycle

Adults (Figure 4.2) are in flight from late April to early June. Eggs are laid in masses at the base of the host plants (Figure 4.3), and hatch after a few days. Larvae spin webs on the host plant and feed communally (Figure 4.4) until reaching fourth instar. In laboratory rearings, this takes approximately six weeks (D. Grosboll, pers. comm.). Upon reaching fourth instar, larvae stop feeding and enter an obligatory period of diapause within the litter or soil. Larvae break diapause in March of the following year (Danby 1890) and spend their time feeding and basking individually until pupation (Figure 4.5).

Figure 4.2. *Euphydryas editha taylori*, adult.

Figure 4.3. Eggs of *Euphydryas editha taylori* in leaf axil of *Castilleja hispida*.

Figure 4.4. Prediapause larvae of *Euphydryas editha taylori* on *Castilleja hispida*.

Figure 4.5. Postdiapause larva of *Euphydryas editha taylori*.

4.3.3. Host plants

Eggs are laid on *Plantago lanceolata* (English plantain) (Llewellyn-Jones 1936), *Castilleja hispida* Benth. (hairy paintbrush) (A. Potter, pers. comm.), and *Collinsia grandiflora* Dougl. (large-flowered blue-eyed Mary) (D. Grosboll, pers. comm.). Larvae feed and develop on all of these species. At sites where *P. lanceolata* and *C. hispida* both occur, eggs have been found on only one species or the other (A. Potter, pers. comm.). Some historic populations occurred at sites where there was no *Castilleja hispida*, but *Castilleja levisecta* Greenm. (golden paintbrush) was present. This was probably also a host plant. Post-diapause larvae feed on *P. lanceolata* and *C. hispida*, as well as *Plectritis congesta* Lindl. (sea blush) (D. Grosboll and A. Potter, pers. comm.). Post-diapause larvae have also been observed on *Plantago maritima* Hult. (seaside plantain) (J. Pelham, pers. comm.). Post-diapause larvae are probably much less discriminating than ovipositing females, and have been reared on a variety of plants in the families Plantaginaceae and Scrophulariaceae.

4.3.4. Nectar sources

Primary nectar sources of extant populations are *Lomatium utriculatum* Coult. and Rose (spring gold), *Plectritis congesta* (sea blush), *Fragaria virginiana* Dushesne (strawberry), and *Camassia quamash* Greene (common camas) (A. Potter, D. Ross, pers. comm.). I have also observed adults nectaring on *Hypochaeris radicata* L. (hairy cat's ear), *Eriophyllum lanatum* Forbes (woolly sunflower), *Sedum spathulifolium* Hook. (broad-leaved stonecrop), *Mahonia aquifolium* Nutt. (tall Oregon-grape), and *Rubus ursinus* Cham. and Schlecht (trailing blackberry).

4.3.5. Sources of mortality

Larvae that have not reached fourth instar by the time the host plants senesce in summer die of starvation. This is likely to be the main source of mortality. Some larvae are killed by an unidentified braconid wasp in one Oregon population, but there is no evidence that this is a major source of mortality. The number of larvae, pupae, and adults that are killed by vertebrate predators is probably low, as the host plants of *E. e. taylori* contain defensive compounds that are believed to make the animal distasteful (Bowers 1981). Invertebrate predators presumably kill some larvae, pupae, and adults.

4.3.6. Dispersal

Little is known of the dispersal habits of *Euphydryas editha taylori*. Dispersal habits of the species as a whole are too varied to make any assumptions. *E. e. taylori* has been observed crossing forested areas, large rivers, and other barriers. Adults have also been found several kilometres from known populations, suggesting at least occasional long-range dispersal (M. Vaughan and S. H. Black, pers. comm.).

4.3.7. Habitat requirements

Cover of *Plantago lanceolata*, the only host plant detected in plots, differed significantly between sites (p< 0.000) (Figure 4.6., Table 4.1.). Cover was significantly higher at Grassy Point, a site that is not known to have ever supported a population of *Euphydryas editha taylori*. Host plant cover at Helliwell Provincial Park, the site most recently occupied, was higher than at the remaining sites, but not significantly so.

Site (N)	Host Plant Cover (%)	Nectar Plant Cover (%)
Gov House (62)	1.4 ± 0.18	0.6 ± 0.07
Grassy Pnt (17)	14.9 ± 3.6*	0.8 ± 0.2
Uplands A (18)	2.7 ± 0.6	6.1 ± 1.4**
Uplands B (22)	1.4 ± 0.3	11.7 ± 2.5*
Tribune Bay (46)	2.5 ± 0.4	0.04 ± 0.006
Helliwell (45)	5.0 ± 0.7	0.07 ± 0.01
ANOVA p=	<0.000	<0.000

Table 4.1. Average cover of host and nectar plants at potential *Euphydryas editha taylori* sites in Victoria and on Hornby Island, BC, with standard error and ANOVA results.

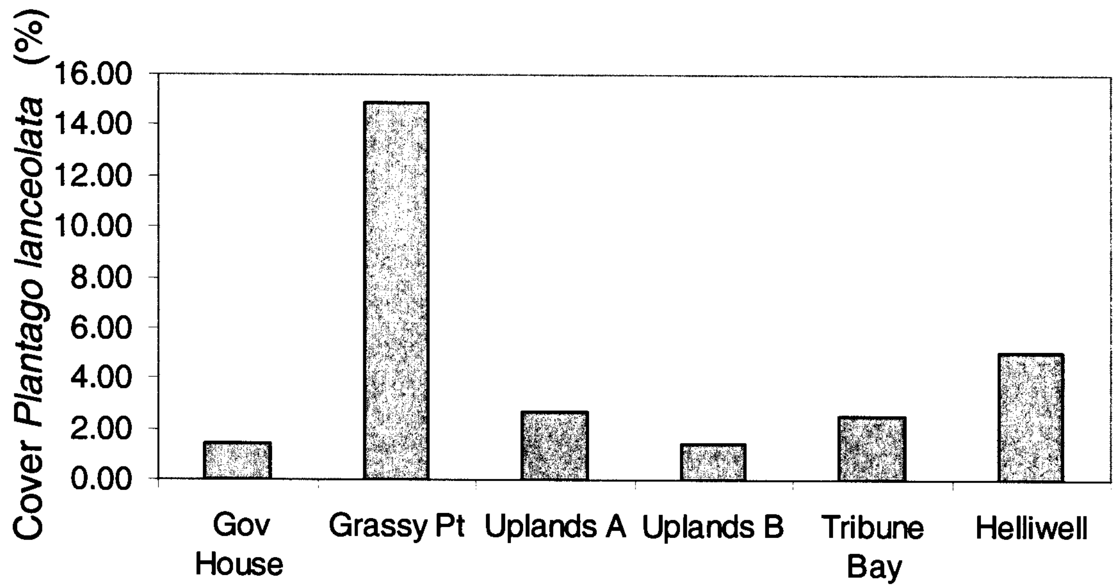

Figure 4.6. Cover of the host plant, *Plantago lanceolata*, at potential *Euphydryas editha taylori* sites in Victoria and on Hornby Island, BC.

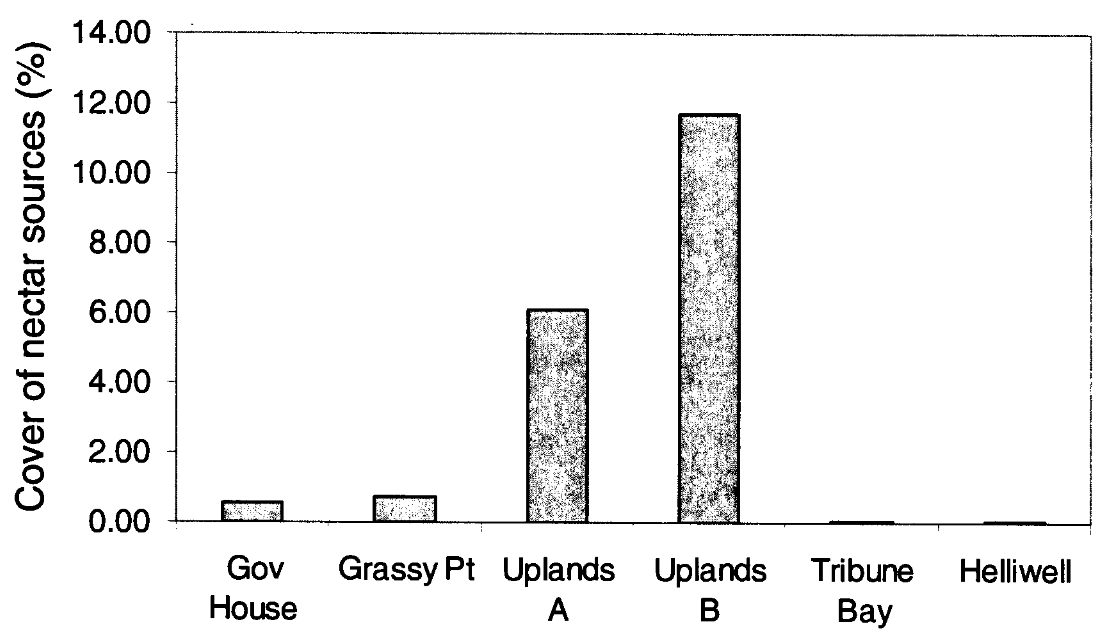

Figure 4.7. Cover of nectar sources at potential *Euphydryas editha taylori* sites in Victoria and on Hornby Island, BC.

Cover of nectar plants differed significantly between sites ($p < 0.000$) (Figure 4.7, Table 4.1). Uplands A had significantly higher cover of nectar sources than any other site except Uplands B. Uplands B had significantly higher cover of nectar sources than all other sites. Sites with the lowest cover of nectar plants were Helliwell Provincial Park and Tribune Bay Provincial Park, the two sites that most recently supported populations of *E. e. taylori*. Cover of nectar plants at Uplands A and B was higher than at Helliwell or Tribune Bay by two orders of magnitude.

In almost all parameters measured, microhabitat around the host plants differed significantly among sites (Figure 4.8, Table 4.2). The only exception was shrub cover ($p = 0.114$). Cover of *Plantago lanceolata* in areas surrounding the host plants was significantly higher at Grassy Point, while other sites were not significantly different from each other. Percentage of unvegetated ground was significantly higher at Grassy Point and Tribune Bay Provincial Park, while the other sites were not significantly different from each other. Unvegetated ground at Grassy Point was mostly rock and exposed soil, while at Tribune Bay Provincial Park it was mostly grass thatch. Cover of short herbaceous vegetation was significantly higher at Uplands A, while other sites were not significantly different from each other. For tall herbaceous vegetation, Uplands B and Grassy Point had significantly lower cover than Tribune Bay Provincial Park, while other sites were not significantly different from each other. Tribune Bay and Helliwell Provincial Parks both had the highest cover of tall herbaceous vegetation, but this was not significant.

For all parameters measured, the recently occupied sites (Helliwell and Tribune Bay Provincial Parks) did not differ significantly from other sites.

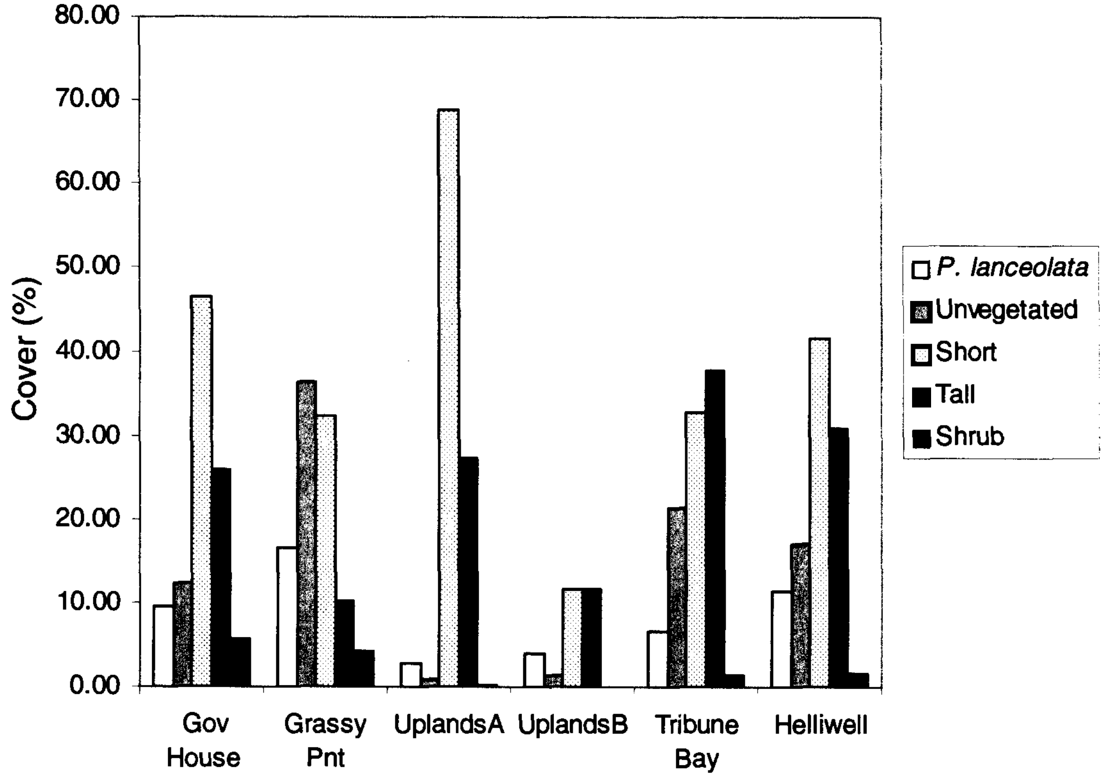

Figure 4.8. Ground cover in 1m^2 plots surrounding individuals of *Plantago lanceolata* at potential *Euphydryas editha taylori* sites in Victoria and on Hornby Island, BC.

Site (N)	*Plantago lanceolata*	Unvegetated	Short	Tall	Shrub
Gov House (20)	9.5 ± 3.1	12.4 ± 2.1	46.4 ± 6.3	25.9 ± 5.7	5.8 ± 1.9
Grassy Pnt (17)	16.8 ± 2.8*	36.4 ± 6.5*	32.5 ± 5.2	10.2 ± 4.4*	4.2 ± 2.9
UplandsA (12)	2.8 ± 1.2	0.9 ± 0.3	68.8 ± 7.4*	27.3 ± 7.4	0.2 ± 0.2
UplandsB (16)	3.9 ± 1.2	1.4 ± 0.5	11.7 ± 4.7	11.7 ± 4.7*	0.06 ± 0.06
Tribune Bay (25)	6.7 ± 1.5	21.5 ± 2.4*	32.8 ± 3.5	37.9 ± 4.2	1.4 ± 0.6
Helliwell (32)	11.4 ± 2.3	17.0 ± 3.7	41.7 ± 3.7	30.9 ± 3.8	1.8 ± 1.4
ANOVA p=	0.002	<0.000	<0.000	<0.000	0.114

Table 4.2. Average cover in 1m^2 plots surrounding individuals of *Plantago lanceolata* at potential *Euphydryas editha taylori* sites in Victoria and on Hornby Island, BC, with standard error and ANOVA results.

Based on the phenology study at Helliwell Provincial Park, individuals of *Plantago lanceolata* began senescing very early in the season (Table 4.3). When tracking was initiated on May 1, all plants were green and turgid (senescence class 1). By May 29, 45% of plants were already in senescence class 2, meaning that they were limp and wilted, but still green. By the middle of June, only 17% of plants were in class 1, and 52% were in class three, meaning that they had begun to turn brown. By July 6, only 7% of plants remained in class one. It was not until mid-August that all plants were in class four, being entirely brown and dry (Table 4.3). Time of senescence was variable and patchy across the study area, and not associated with visible gradients in the landscape (Figure 4.9).

Observation Date	Class 1	Class 2	Class 3	Class 4
May 1	100%	0	0	0
May 29	45%	55%	0	0
June 16	17%	31%	52%	0
July 7	7%	15%	66%	12%
August 14	0	0	0	100%

Table 4.3. Percentage of individuals of *Plantago lanceolata* in different senescence classes at five observation dates in 2003 at Helliwell Provincial Park (n = 40).

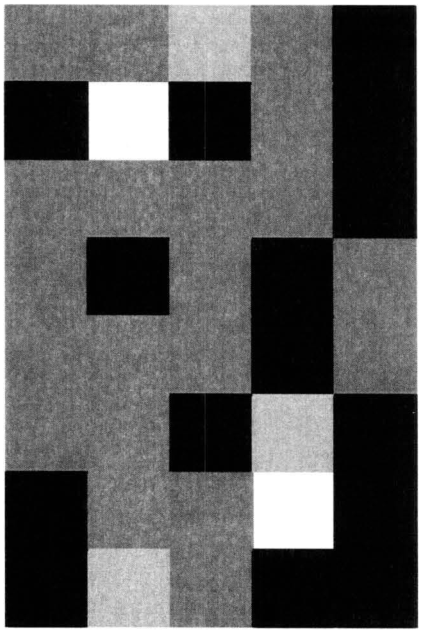

Figure 4.9. Time of senescence of *Plantago lanceolata* across the landscape at Helliwell Provincial Park, 2003. Cells correspond to 10x10m squares with one plant in the centre. Darker cells represent plants that senesced earlier. Black squares represent plants that were senescent on May 29, the earliest date of senescence. White squares represent plants that were not yet senescent on July 6, the penultimate observation date.

4. 4. <u>Discussion</u>

Euphydryas editha uses a variety of host plants, mostly from the families Scrophulariaceae and Plantaginaceae (White and Singer 1974). Common to all host plants is the presence of iridoid glycosides as defensive compounds (Bowers 1983). These compounds are thought to discourage generalist herbivores, but stimulate feeding in butterflies of the genus *Euphydryas* (Bowers 1983).

In recent times, several subspecies of *Euphydryas editha* have successfully adapted to new host plants. In the eastern Sierra Nevada range, where *Plantago lanceolata* has been introduced through cattle ranching, *E. editha* has adopted this plant as a host, in addition to *Collinsia parviflora* and *Castilleja* spp. (Thomas *et al.* 1987, Singer *et al.* 1993). This range of hosts is very similar to that used by *E. e. taylori*. In the western Sierra Nevada, populations that exclusively used *Pedicularis semibarbata* in the past have recently adopted a new host, *Collinsia torreyi*. This new host has been adopted only in areas that have been logged and burned, where the plants of *C. torreyi* grow larger and in greater abundance than in the surrounding area (Singer *et al.* 1992). In both of these cases, the butterflies have adapted to anthropogenic changes to their immediate surroundings, but have not expanded their range to take advantage of similar anthropogenic habitat. This also appears to be true of *E. e. taylori*, which has adopted *Plantago lanceolata* as a host plant as it has invaded the butterfly's habitat. Adaptation to a novel host raises several questions relevant to the conservation of this taxon. Do females oviposit preferentially on the native or the introduced host? Does larval development and survival differ between hosts? These questions have been addressed elsewhere in the range of *Euphydryas editha*.

In populations that do not have access to *Plantago lanceolata*, some females will nonetheless oviposit on it readily if it is offered (Thomas *et al.* 1987, Singer *et al.* 1993). Those populations that do have access to *Plantago lanceolata* contain some females that will oviposit on it preferentially, choosing the ancestral host only when *P. lanceolata* is not available. The proportion of females that prefer *P. lanceolata* is increasing in these populations, and some females will not oviposit on the ancestral host even when nothing else is available (Singer *et al.* 1993). This preference has been shown to have a genetic basis, meaning that the population is evolving a preference for the novel host (Singer 1988 *et al.*, Singer *et al.* 1993; Thomas *et al.* 1987). The same process is happening where populations have adopted *Collinsia torreyi* (Singer *et al.* 1993). These results suggest that populations of *Euphydryas editha taylori* that have adopted *Plantago lanceolata* as a host may reject the ancestral hosts. This has implications in any future reintroduction attempts. Care must be taken to choose a donor population with similar host plants to the reintroduction site. It is interesting to note, however, that no population of *E. e. taylori* has yet been found to oviposit on both *Plantago lanceolata* and *Castilleja hispida*, although both plants are present at several sites (A. Potter and D. Grosboll, per. comm.).

Comparisons of larval performance on different hosts have produced conflicting results. Some studies have shown little difference in larval performance on novel and ancestral hosts. Other studies have shown higher larval growth and survival rates on the novel host (Moore 1989). Still other studies have shown that there is variation between individuals of the same population, and that some larvae perform better on the novel host, and some on the ancestral. In these cases, females oviposit preferentially on the host that confers the highest growth rate to their offspring (Singer *et al.* 1988). They also tend to remain in areas containing the preferred host, and to disperse from areas containing only alternate hosts (Thomas and Singer 1987). In coastal California, where *Euphydryas editha* larvae feed on either the native *Plantago erecta* or *Castilleja* spp., it has been shown that *Castilleja* spp. are more suitable hosts, both because they confer higher larval growth rates, and because they senesce later, increasing larval survival (Singer 1972, Hellmann 2002). However, when *E. e. taylori* is reared in the lab, both survival and growth rate are equal on *Plantago lanceolata* or *Castilleja hispida* (D. Grosboll, pers. comm.). The relative suitability of the different host plants of *E. e. taylori* should be investigated further in the field, in order to determine how important each host might be in population persistence. If one host confers significantly higher larval survival, its presence or absence could be a major factor in determining the suitability of a reintroduction site.

Females of *Euphydryas editha* emerge from the chrysalis with mature eggs, and can live a short time and oviposit without feeding on nectar. However, females that have access to nectar live longer, and lay more eggs, particularly in those egg masses that are laid later in the life of the butterfly (Murphy *et al.* 1983). These later egg masses are unlikely to produce viable offspring, because of host plant senescence. However, in years when wet weather delays host plant senescence, these later egg masses may allow a population to increase in size, buffering against future stochastic events. If nectar is important to population persistence, then the diversity of nectar plants may be important, so that nectar is available through the duration of the flight season, throughout the available habitat, and in times of climatic extremes. Risk of extinction has been shown to increase with decreasing nectar plant diversity (Murphy *et al.* 1983). In the closely related *Euphydryas chalcedona*, females oviposit preferentially on those host plants that grow in close proximity to nectar sources (Murphy 1983, Murphy *et al.* 1984). This is important to the conservation of *E. e. taylori*, because at sites where nectar sources are rare or patchy, they may not be growing close to those host plants that are most suitable for larval development. This may cause females to oviposit on unsuitable hosts.

Understanding the dispersal abilities and population dynamics of *Euphydryas editha* may be critical to successful conservation planning for *E. e. taylori*. For the species as a whole, dispersal abilities differ greatly between different groups of populations. Those inhabiting low elevation grasslands in western California are known to be extremely sedentary, rarely crossing even open grassland areas that are adjacent to their breeding habitat (Ehrlich 1961, Ehrlich 1965). However, a small proportion of individuals may occasionally disperse many kilometres and found new populations (Harrison *et al.* 1988, Harrison 1989). Presently, these butterflies persist in a metapopulation analogous to a mainland and islands, where a large central population is robust to natural perturbations, and smaller satellite populations undergo frequent extinctions and recolonizations (Harrison *et al.* 1988). In contrast to this situation, montane populations of *Euphydryas editha* have not been shown to have mainland-and-islands dynamics. Studied populations show complex source-sink dynamics, where a few small but persistent populations are scattered throughout a series of larger, but more extinction-prone, populations (Boughton 1999). When the extinction-prone populations are large, the persistent populations behave as sinks, but the roles are reversed when the extinction-prone populations crash (Thomas *et al.* 1996). Butterflies in this system are comparatively strong dispersers, and readily cross forested areas, shrublands, and other non-habitat (Gilbert and Singer 1973).

The more sedentary *Euphydryas editha* populations are found in stable climax communities, while the strongly dispersing populations occur in early successional habitats (Gilbert and Singer 1973). In addition, the sedentary populations occur in structurally simple grassland habitats, while the strongly dispersing populations occur in a matrix of meadow, shrubland, and coniferous forest. The habitats occupied by *E. e. taylori* share characteristics with both climax and early successional habitats. Like early successional habitats, sites are typically small, and isolated from each other by barriers such as forest. This may favour strong dispersers that can cross barriers and colonize new areas. However, in the context of the historical disturbance regime, the habitats were climax communities. There may not have been any selection pressure to produce strong dispersers if habitat were stable.

Observations of *Euphydryas editha taylori* indicate that long distance dispersal does occur, at least occasionally, and that the butterflies do cross forested areas and other barriers. However, anthropogenic habitats have rarely been colonized, and many populations appear to have persisted in relative isolation for long periods. It seems most likely that this subspecies is a poor disperser and colonizer, like the coastal Californian subspecies, to which it is ecologically similar. There is some evidence that *E. e. taylori* may sometimes form mainland and islands metapopulations. Two groups of populations in Oregon consist of a large population surrounded by smaller populations, and a similar group of populations once existed on Hornby Island, BC. The importance of metapopulation dynamics to subspecies *taylori* is unknown. However, based on studies of other subspecies, it is reasonable to assume that multiple populations in close proximity will be more likely to persist in the long term.

While there have been many documented extinctions *Euphydryas editha* populations, few have resulted directly from a decline in the abundance of host plants (Weiss 1999). It is therefore not surprising that, in this study, the abundance of host plants in formerly occupied sites was within the range of variability of sites that have never been occupied. While a decline in nectar sources has been suggested as a cause of population extinction (Shepard 2000, Guppy and Shepard 2001), and low nectar source diversity may predispose populations to extinction (Murphy *et al.* 1983), no population extinction has ever been shown to have been caused by declines in nectar sources. It is therefore not surprising that, in this study, the abundance of nectar sources at formerly occupied sites was highly variable, and the most recently occupied sites had the lowest cover of nectar sources. These results show that the extirpation of *Euphydryas editha taylori* from British Columbia cannot be explained by declines in the abundance of host plants or nectar sources.

Formerly occupied sites were within the range of variability of sites that have never been occupied with respect to percent cover of host plants, bare ground, and short herbaceous vegetation in the immediate area surrounding host plants. These parameters do not appear to be good indicators of habitat quality, and do not explain the absence of *Euphydryas editha taylori* in recent times. Cover of shrubs in the area surrounding the host plants was lower, though not significantly so, at sites that were formerly occupied than at sites that have never been occupied. This may be important, as it has been shown that *Euphydryas editha* larvae that develop under a partial shrub canopy have slowed growth and development, and larvae in shrubby habitats seek the least shaded microsites (Osborne and Redak 2000). In addition, most extant populations of *E. e. taylori* occur in habitats with little shrub cover, and the few egg masses and early instar larvae that have been found were in open areas, away from shrubs. This conflicts somewhat with the observation that the most recently occupied sites had the highest cover of tall herbaceous vegetation. While tall herbaceous vegetation may reduce habitat quality due to shading, it may also be an indicator of a more mesic, and therefore higher quality, site. This paradox has been noted for *Melitaea cinxia* (Glanville fritillary), a related species that also feeds on *Plantago lanceolata*, and is also sensitive to drought (Hanski *et al.* 1996). The significance of moisture regime is discussed below.

Monitoring of host plant phenology at Helliwell Provincial Park showed that the site is presently too dry to be considered high quality habitat. Observations of larvae of *E. e. taylori* in the field and in the laboratory show that larvae feed only on fresh new growth of *P. lanceolata* (Danby 1890, D. Grosboll, pers. comm.). This means that only those plants in senescence class 1 would be acceptable to larvae. By the end of May, approximately half the *P. lanceolata* plants monitored were drought-stressed, and in the early stages of senescence. Since larvae must feed for approximately six weeks, and the flight season begins in late April, even the first females to oviposit have only a 50% chance of producing offspring, if eggs are laid randomly. Females emerging near the end of the flight season, in early June, have a much lower chance. Less than 7% of host plants would still be acceptable to larvae six weeks later. Time of host plant senescence is of critical importance, as larval starvation caused by host plant senescence is believed to be the main cause of mortality, and the extinction of the Hornby Island populations is believed to have been caused by an unusually dry summer in 1998 (Guppy and Fischer 2001).

Drought is a major cause of population extinctions among ecologically similar subspecies of *Euphydryas editha* in California (Singer and Ehrlich 1979, Ehrlich *et al.* 1980, Dobkin *et al.* 1987, Murphy and Weiss 1988). One study found that a maximum of 20% of eggs laid by females emerging on the first day of the flight season would produce larvae that reach diapause, and that this figure declined to 5% only one week into the flight season (Cushman *et al.* 1994). Larval mortality due to host plant senescence exceeds 80% in an average year (Dobkin *et al.* 1987).

High quality habitat is not necessarily the same for prediapause and postdiapause larvae. In the well-studied populations of *Euphydryas editha* in California, time of host plant senescence varies across the landscape, due to the effects of topography on microclimate (Dobkin *et al.* 1987, Weiss *et al.* 1988). However, microclimate also affects the rate of development of postdiapause larvae. Those individuals that develop in warmer microsites will eclose earlier, and therefore have a higher probability of producing offspring that will survive to diapause. However, postdiapause larvae suffer higher mortality in warmer microsites, because of advanced host plant phenology (i.e. earlier senescence). Those individuals that develop in cooler microsites have higher survival rates, but eclose later, and are less likely to produce offspring that will survive (Dobkin *et al.* 1987). This does not mean that no site can be ideal for both however, because postdiapause larvae can disperse up to 10m per day to find a favourable microsite (Weiss *et al.* 1987). High quality habitat for a population, then, must feature topographic and microclimatic heterogeneity on a scale of tens of meters, so that microsites ideal for postdiapause larvae are closely juxtaposed with microsites for prediapause larvae (Weiss *et al.* 1987, Weiss *et al.* 1988).

Today the sites formerly occupied by *Euphydryas editha taylori* are topographically simple, at least at a meso-scale. Despite this apparent simplicity, the phenology of *Plantago lanceolata* is variable across the landscape at Helliwell Provincial Park (Figure 4.9.). Thus moisture regimes, and possibly microhabitats, vary across the landscape. However, there has been extensive woody encroachment into open areas at Helliwell Provincial Park, as well as at other formerly occupied sites in BC (Penn and Dunster 2001, Collier *et al.* 2004). Because woody encroachment is fastest on more mesic and deep-soiled sites, in depressions and on cooler aspects, this encroachment has reduced the topographic, hydrological, and microclimatic heterogeneity of the site, and, therefore, reduced its quality as butterfly habitat.

The case of *Euphydryas editha taylori* in British Columbia appears to be similar to that of many other rare butterflies in areas that have long histories of habitat management. It has been repeatedly shown that, for these butterflies, habitat quality is much more important than quantity, and that the maintenance of this quality depends on active management.

The major cause of historic declines of *E. e. taylori* was habitat destruction. In recent times, the major cause of decline appears to be the deterioration of habitat quality because of woody encroachment, which is especially problematic in combination with extreme climatic events. Extreme events are predicted to become more frequent with climate change, and this increasing climatic variability may have already hastened the extinction of some *Euphydryas editha* populations in California (McLaughlin *et al.* 2002). However, there is no evidence that climate change has been a factor in the extirpation of *E. e. taylori* from British Columbia. Climate change has been reported as a possible factor in British Columbian extinctions (Parmesan 1996). However, in that case, the role of extreme habitat alteration was ignored. There is no reason to conclude that *Euphydryas editha taylori* could not be reintroduced to British Columbia, provided that former habitats are restored, and that they are monitored and managed.

4. 5. References

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5. Removal of conifers to restore butterfly habitat in grasslands of southwestern British Columbia: A case study

5.1. Introduction

Before European colonization, North America contained a variety of ecosystems that had an open structure, maintained by frequent fire (Thilenius 1968, Grimm 1984, Anderson and Bailey 1980, Noss 1989, Samson and Knopf 1994, Agee 1996). Following European colonization and subsequent fire suppression, many of these open ecosystems began to fill in with trees and shrubs. The replacement of herbaceous communities by woody vegetation has become an issue of conservation concern throughout North America, as those species that rely on open plant communities continue to decline (del Moral and Deardorff 1976, White 1983, Agee and Dunnwiddie 1984, Hardin 1988, Grigore and Tramer 1996, Kurczewski 1998, McCarty 1998, Tveten and Fonda 1999). There have been many attempts to restore the open structure of these degraded ecosystems, many of which have involved the reintroduction of fire. However, fire can also have undesirable effects. Many plant communities are now so changed that simple reintroduction of fire will not suffice to restore historic structure or composition. Fuel loads may have accumulated to the point that fires will burn too hot and kill too many trees or remove too much soil, or plant communities could be so changed that desired species cannot reinvade, because no propagule sources exist. In addition, interactions between exotic species and fire can result in a highly undesirable plant community following fire. Also, fire may be looked upon unfavourably by the public and by policy makers. In this context, it is prudent to experiment with surrogates for fire.

Before Euro-Canadian colonization, the Garry oak ecosystem of southwestern British Columbia was characterized by frequent, low intensity fires (Fuchs 2001). These fires were set intentionally by indigenous people as part of the management of important vegetable foods. Fire suppression in the Garry oak ecosystem began around 1850, and has increased in effectiveness ever since. Today, many mesic sites have been converted from open oak parkland to coniferous forest, closed oak forest, or dense shrub thickets. Over one hundred species at risk are known from the Garry oak ecosystem, and almost all are associated with meadows or open woodland. While restoration efforts in the Garry oak ecosystem have largely focussed on the removal of non-native shrubs, there is increasing concern about the expansion of native shrubs and conifers into formerly open areas.

Butterflies have been heavily impacted by woody encroachment. Of the thirteen butterflies at risk that are known from the Garry oak ecosystem, at least five are associated with mesic or wet meadows (Guppy and Shepard 2001). These habitats have largely been lost to woody encroachment, invasive species, and development. One species, Taylor's checkerspot, *Euphydryas editha taylori* Edwards, is now threatened throughout its range by forest encroachment (Vaughn and Black 2002). Studies of former Taylor's checkerspot habitat in Helliwell Provincial Park on Hornby Island, BC, showed that forest encroachment has reduced the area of available habitat, and reduced the fine scale heterogeneity of the site (Chapter 4, this thesis). In order to restore the site for a reintroduction attempt, conifers will have to be removed. Because the trees are well established and have continuous canopies, the use of fire would be dangerous and controversial. Accordingly, restoration will have to involve mechanical removal of trees. However, it is not known how the plant community will respond to large-scale removal of encroaching conifers. The purpose of this study was to document the encroachment of conifers into grassland areas, and to describe changes to the plant community after experimental removal of a small number of trees.

5.2. Methods

5.2.1. Study site

Helliwell Provincial Park is located on Hornby Island in the Straight of Georgia, BC (Figure 5.1.). The park consists of a 2803 ha marine portion and a 69 ha terrestrial portion. The terrestrial portion of the park includes forests of various ages and compositions, coastal bluffs, and grassland. Grasslands occur mainly as a strip between the forests and coastal bluffs on south-facing areas. The grasslands are dominated by native and introduced graminoids such as *Danthonia californica* Boland. (California oatgrass), *Carex inops* Bailey (long-stoloned sedge), *Bromus* spp. (brome), *Agrostis* spp. (bentgrass), and *Anthoxanthum odoratum* L. (sweet vernalgrass), as well as bryophytes. The grasslands produce an impressive display of native flowers in the spring, including *Zygadenus venenosus* S. Wats. (meadow death-camas), *Lomatium utriculatum* Coult. and Rose (spring gold), *Ranunculus occidentalis* Nutt. (western buttercup), *Delphinium menz*iesii DC (Menzies' larkspur), *Cerastium arvense* L. (field chickweed), and *Eriophyllum lanatum* Forbes (woolly sunflower).

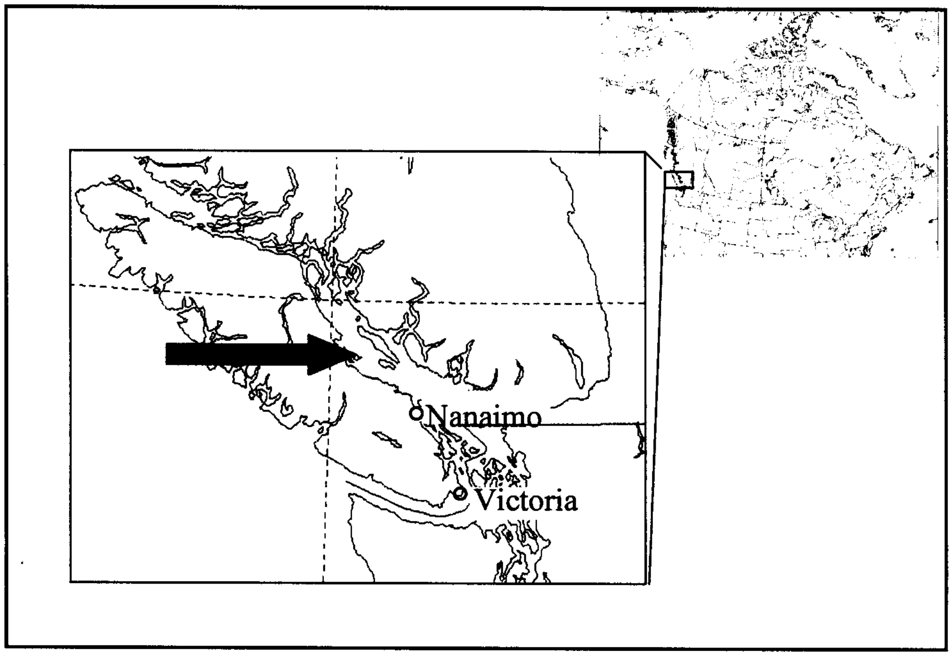

Figure 5.1. Location of Hornby Island.

It is believed that these grasslands were formerly burned by First Peoples as part of the management of various root vegetables (Penn and Dunster 2001). Casual observations by local naturalists have been warning of forest encroachment into the grasslands for many years. Tree species expanding into grassland areas are *Arbutus menziesii* Pursh. (arbutus), *Pinus contorta* Dougl. (shore pine), and especially *Pseudotsuga menziesii* Franco (Douglas-fir). The largest areas of grassland occur in the western-most part of the park. This area was occupied by Canada's last known population of Taylor's checkerspot, *Euphydryas editha taylori*, as recently as 1995. Surveys of the park in 2001 and 2003 both concluded that the butterfly is now extirpated (Guppy and Fischer 2001, Miskelly 2003).

5.2.2. Documenting encroachment

Position and age of Douglas-fir trees were recorded throughout the western-most 240 m of Helliwell Provincial Park, the area formerly occupied by Taylor's checkerspot. This was an area of approximately 3 ha. The northern border of the park was used as a baseline, from which 1 m-wide belt transects were run from north to south every 20 m. The transects ended where they met a trail that follows the top of the coastal bluffs. The position of every tree trunk that was found in these transects was recorded. If the tree was less than 3 m tall, the height was recorded. If the tree was more than 3 m tall, the diameter at breast height (1.3 m) was recorded.

Ages were estimated for a subsample of trees. For trees less than 3m tall (subsample N= 32), this was done by counting internodes between whorls of branches. For trees more than 3 m tall (subsample N= 17), this was done using an increment borer to extract a core at 30 cm above ground. Tree rings on the cores were counted to determine age. Ages determined were considered minima. The actual age of the trees is assumed somewhat greater. This is because coring trees at a height of 30 cm fails to account for the time taken to reach this height, and because many young trees at Helliwell are heavily browsed by deer, leading to several years of stunted and disorganized growth that makes aging the trees difficult. Data from coring and measuring trees were used to create DBH versus age and height versus age plots for the two size classes of trees. Diameter, height, and minimum age were all log transformed to improve fit. Formulae derived from these plots were used to determine minimum age of all trees recorded in the belt transects. Position and minimum age of all trees were plotted to illustrate the distribution of different age classes of trees across the grassland.

5.2.3. Vegetation response to removal of conifers

In February 2003, ten groups of invading Douglas-fir trees containing one to three trees each were cut at ground level and removed from the grassland. In April and July 2003 and June 2004, the plant community in the areas from which trees had been removed was monitored. From the centre of each area that had formerly been shaded by a standing tree, four radii were placed. One radius was placed randomly in each of four quadrants. Along these radii, the ground cover was recorded on a per centimetre basis. Vascular plants were identified to species and other ground cover was classified as lichen, moss, rock, thatch, duff, or bare soil. For each plot treated, two plots were also monitored as controls. One plot was the closest possible area of the same size that was still under the canopy of standing trees. In most cases, this was immediately adjacent to the treated area. The second control was an area of grassland of the same size as the treated area. This area was chosen to be most similar to the treated area in terms of aspect and surrounding vegetation, and was never more than 6m from the treated area.

Data for ground cover along each radius were summed for each plot and converted to percentages. Ground cover was combined into functional groups (unvegetated ground; native and exotic forbs, graminoids, and shrubs; total native; total exotic) and treated plots were compared to both controls using ANOVA. Arcsine transformation of data to improve normality did not affect ANOVA results, and so is not presented. Significant differences were determined using Scheffe's Test. These statistics were calculated using the program SPSS 12. In addition, all plots were compared using multidimensional scaling. This technique reduces multiple variables (in this case, ground cover by species) to two new derived variables, which can then be plotted in twodimensional space (Quinn and Keough 2002). Multidimensional scaling was done using the program Statistica 6. Nomenclature of plants follows Douglas *et al.* (2001).

5.3. Results

5.3.1. Ages and distribution of encroaching trees

Age of Douglas-fir trees was well correlated with height and diameter at breast height (Figure 5.2, Figure 5.3). However, the ages calculated using the formulae from these plots must be considered estimates, as there is considerable spread around the trend lines.

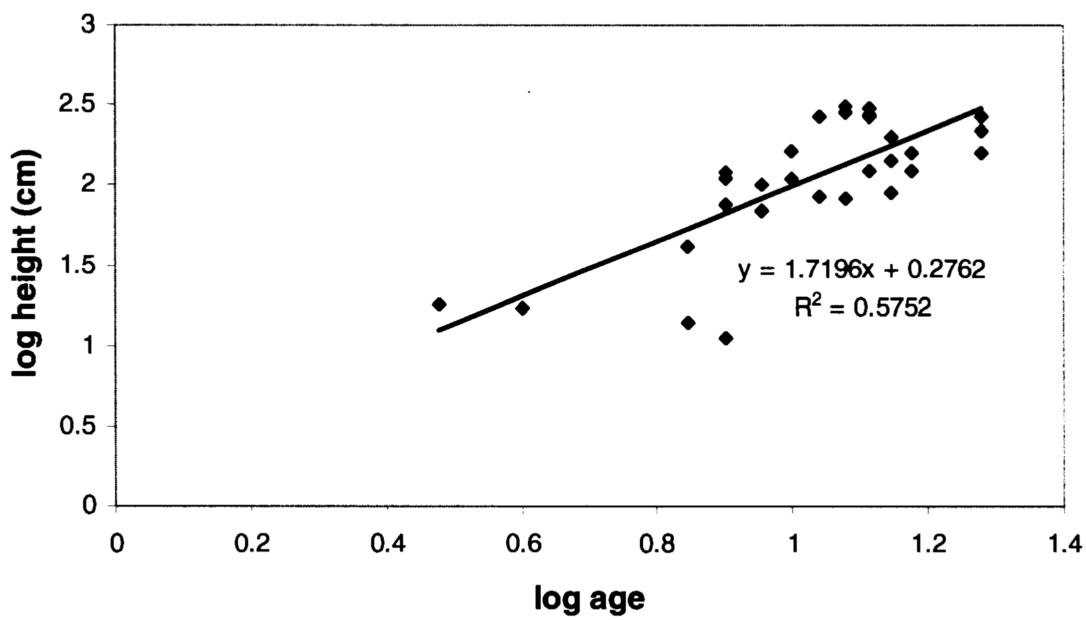

Figure 5.2. Logarithm of minimum age in years versus logarithm of height in cm for 32 Douglas-fir trees growing in grasslands at Helliwell Provincial Park.

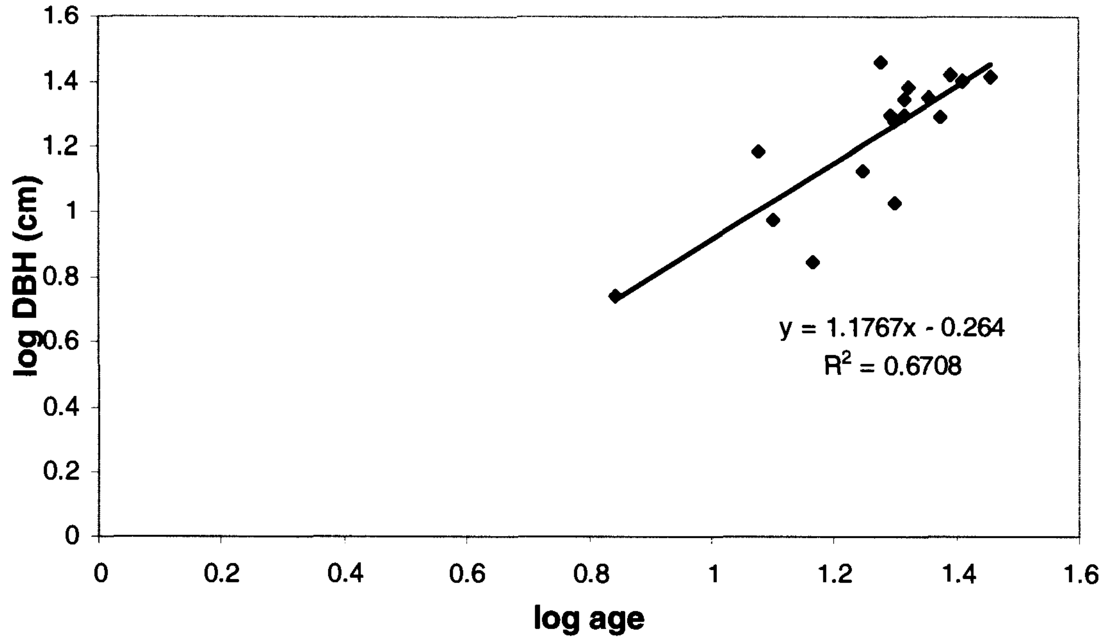

Figure 5.3. Logarithm of minimum age in years versus logarithm of diameter at breast height in cm for 17 Douglas-fir trees growing in grasslands at Helliwell Provincial Park.

Average minimum age of Douglas-fir trees in the area sampled was 12.5 years (N = 92, Standard Error = 0.74, Median = 11.8 years). The oldest tree had a minimum age of 44 years. The majority of trees had a minimum age of 10-15 years (Figure 5.4). Older trees (minimum age > 20 years) were found mainly in depressions, at the base of slopes, and towards the baseline of the transects (i.e. further from the bluffs and closer to continuous forest) (Figure 5.5). Younger trees (minimum age < 20 years) occurred as scattered individuals throughout the grassland, but were most abundant in the same areas as the older trees (Figure 5.5).

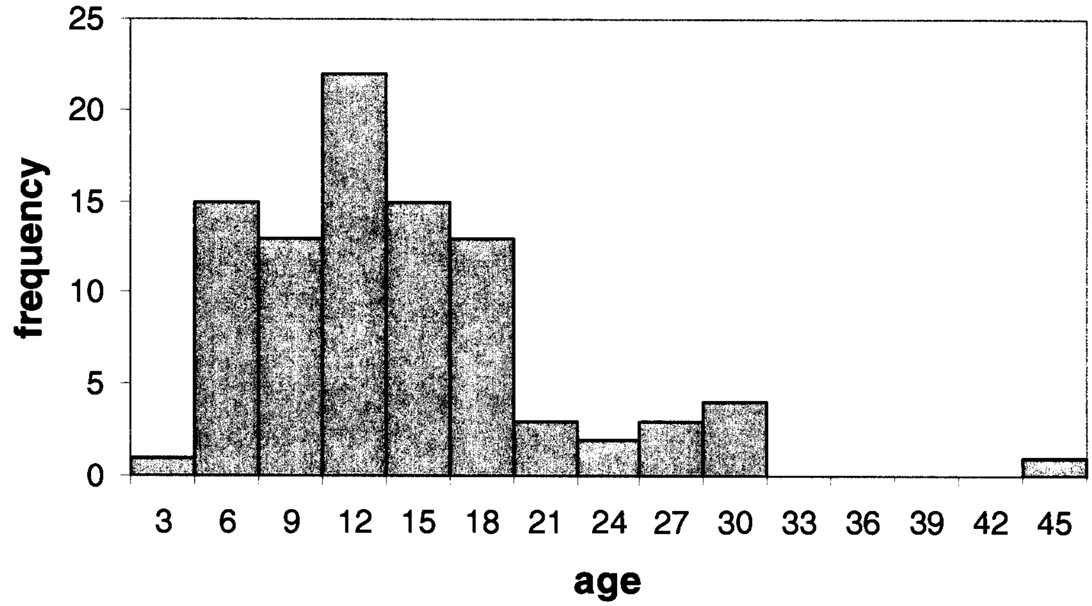

Figure 5.4. Histogram of ages of Douglas-fir trees growing in grasslands at Helliwell Provincial Park.

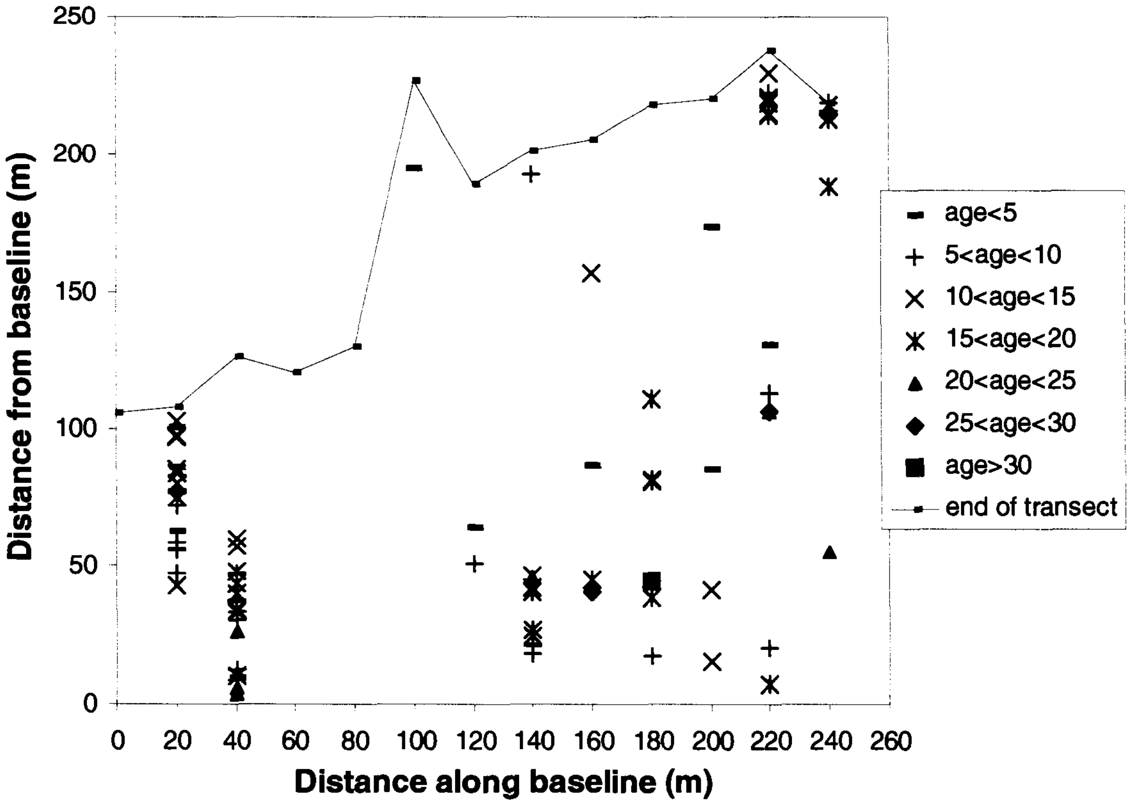

Figure 5.5. Positions of Douglas-fir trees of different age classes recorded on transects at 20m intervals in grasslands at Helliwell Provincial Park

5.3.2. Vegetation response to conifer removal

Following removal of trees, the plots initially appeared almost devoid of vegetation (Figure 5.6). In the first growing season following tree removal, many perennial grassland species were observed in the treated areas. These plants were too mature to have grown from seed recently, and must have been surviving in a suppressed state under the shade of the trees. Some of these plants flowered in the first year of release. In the second year of release, more grassland species were noted in the treated plots and more individuals flowered. Some species that were well represented in the treated plots were *Brodiaea coronaria* Engl. (harvest brodiaea) (Figure 5.7), *Triteleia hyacinthina* Greene (fool's onion), *Ranunculus occidentalis* (western buttercup), *Perideridia gairdneri* Mathias (Gairdner's yampah), and *Pteridium aquilinum* Kuhn (bracken fern). Graminoids that were commonly seen in treated plots included *Carex inops* (long-stoloned sedge), *Festuca rubra* L. (red fescue), and *Elymus glaucus* Buckl. (blue wildrye) (Figure 5.8). Only one native species was observed growing from seed in treated areas, *Dichanthelium oligosanthes* Gould (Scribner's witchgrass) (Figure 5.9).

Some exotic species also survived under the shade of the canopy, and grew rapidly following release. These included *Dactylis glomerata* L. (orchard-grass) (Figure 5.10), *Poa* spp. (bluegrass), and *Rumex acetosella* L. (sheep sorrel). Many other exotic species rapidly colonized the bare soil that dominated the treated areas in the first season following tree removal. These included a variety of annual forbs and grasses, and perennial species such as *Cirsium vulgare* Tenore (bull thistle) (Figure 5.11), *Vicia sativa* L. (common vetch), and *Holcus lanatus* L. (common velvet-grass).

Figure 5.6. A typical treated plot immediately after treatment in 2003.

Figure 5.7. *Brodiaea coronaria*, harvest brodiaea, flowering in a treated plot in 2004.

Figure 5.8. *Elymus glaucus*, blue wildrye, growing in a treated plot in 2004.

Figure 5.9. Seedlings of *Dichanthelium oligosanthes*, Scribner's witchgrass, growing in a treated plot in 2003.

Figure 5.10. *Dactylis glomerata*, orchard-grass, growing in a treated plot in 2003.

Figure 5.11. *Cirsium vulgare*, bull thistle, growing in a treated plot in 2004.

After one growing season, many plots were still largely unvegetated (Figure 5.12). However, after two growing seasons, the treated areas had been colonized and were difficult to recognize within the surrounding grassland (Figure 5.13). After two growing seasons, treated plots did not differ significantly from controls in the grassland in any parameter measured: cover of native and exotic graminoids, cover of native and exotic forbs, cover of native or exotic shrubs, or bare ground. Treated plots differed significantly from tree-covered control plots only in terms of bare ground, which was lower in treated plots. Grassland control and tree-covered control plots differed from each other in the cover of native graminoids, total cover of native plants, and the extent of bare ground (Figure 5.14, Table 5.1, Table 5.2).

Figure 5.12. A typical treated plot in 2003, following treatment and one growing season.

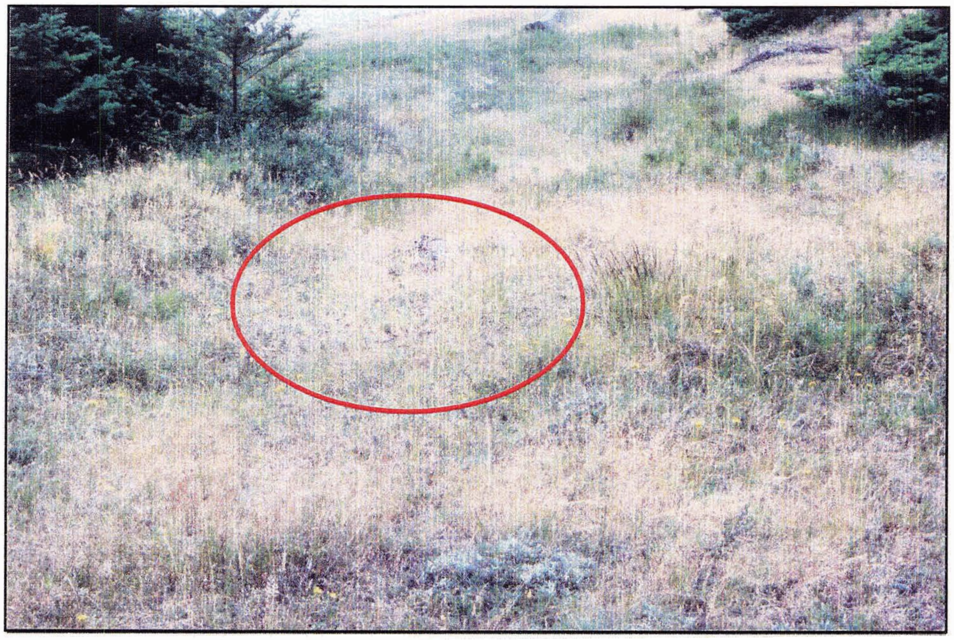

Figure 5.13. A typical treated plot in 2004, two growing seasons after treatment. Circle indicates treated area.

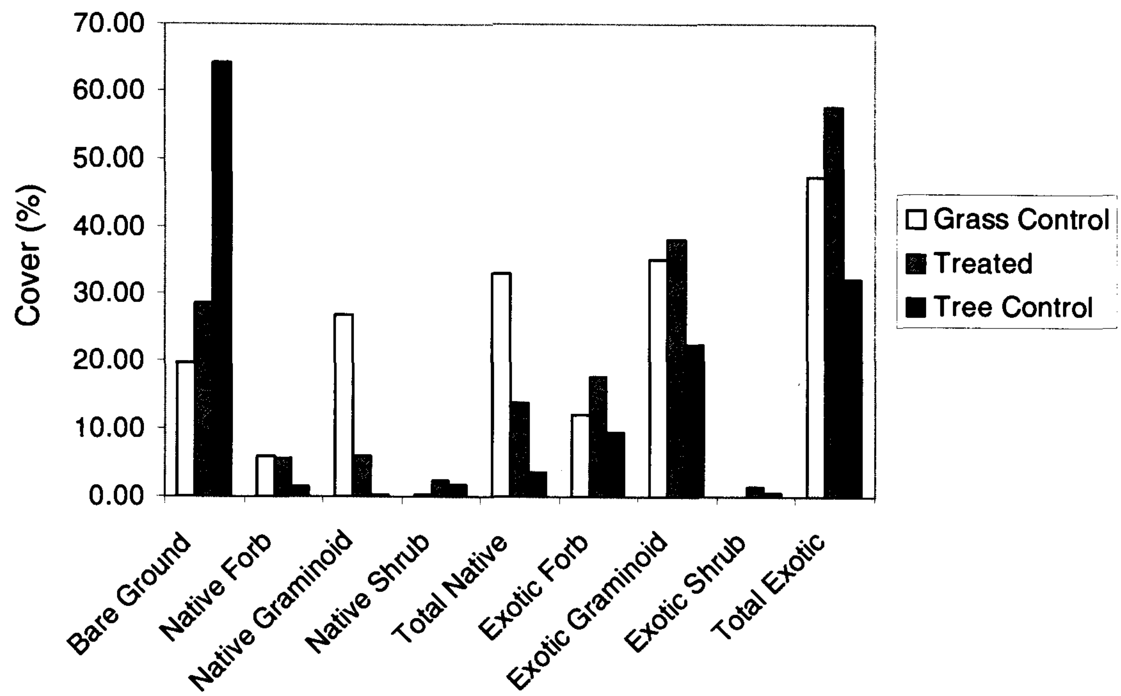

Figure 5.14. Ground cover of treated and untreated plots in June 2004, two growing seasons after treatment.

Treatment	Treated	Grass Control	Tree Control	ANOVA p=
Unvegetated	28.7 ± 7.8	19.7 ± 7.6	64.2 ± 8.9*	0.001
Native Forb	5.6 ± 3.2	5.8 ± 3.7	1.5 ± 1.3	0.514
Native Graminoid	5.8 ± 3.3	26.9 ± 7.1*	0.2 ± 0.2**	0.001
Native Shrub	2.5 ± 1.8	0.2 ± 0.2	1.8 ± 1.6	0.515
Total Native	13.9 ± 4.5	33.0 ± 7.6*	3.4 ± 2.0**	0.002
Exotic Forb	17.7 ± 2.6	12.1 ± 2.0	9.4 ± 4.5	0.191
Exotic Graminoid	38.2 ± 5.0	35.3 ± 7.6	22.4 ± 5.4	0.172
Exotic Shrub	1.6 ± 1.6	0.0 ± 0.0	0.6 ± 0.6	0.510
Total Exotic	57.5 ± 6.5	47.4 ± 7.5	32.3 ± 8.7	0.074

Table 5.1. Ground cover of treated and untreated plots at Helliwell Provincial Park in June 2004, two growing seasons after treatment, with standard error and ANOVA results.

Despite the similarity of cover in the treated and grassland control sites, multidimensional scaling shows that the sites are still quite different in terms of the species present and their abundances. Figure 5.15 shows that treated, grassland control, and tree-covered control plots are generally separate from each other. Points on the graph that are closer together represent plots that are more similar in ground cover at the species level. Tree-control plots cluster most closely, while grassland control plots show the greatest dispersion. Two tree-covered control plots (TC7 and TC8) occur among the treated plots, while one treated plot (T7) occurs among the grassland control plots.

Differences between treated and grassland control plots are primarily due to the colonization of treated plots by exotic species and the lower cover of native graminoids in treated plots. Figure 5.16 shows that the major change in the treated plots between 2003 and 2004 was the colonization of unvegetated ground by exotic species. The cover of native species changed very little.

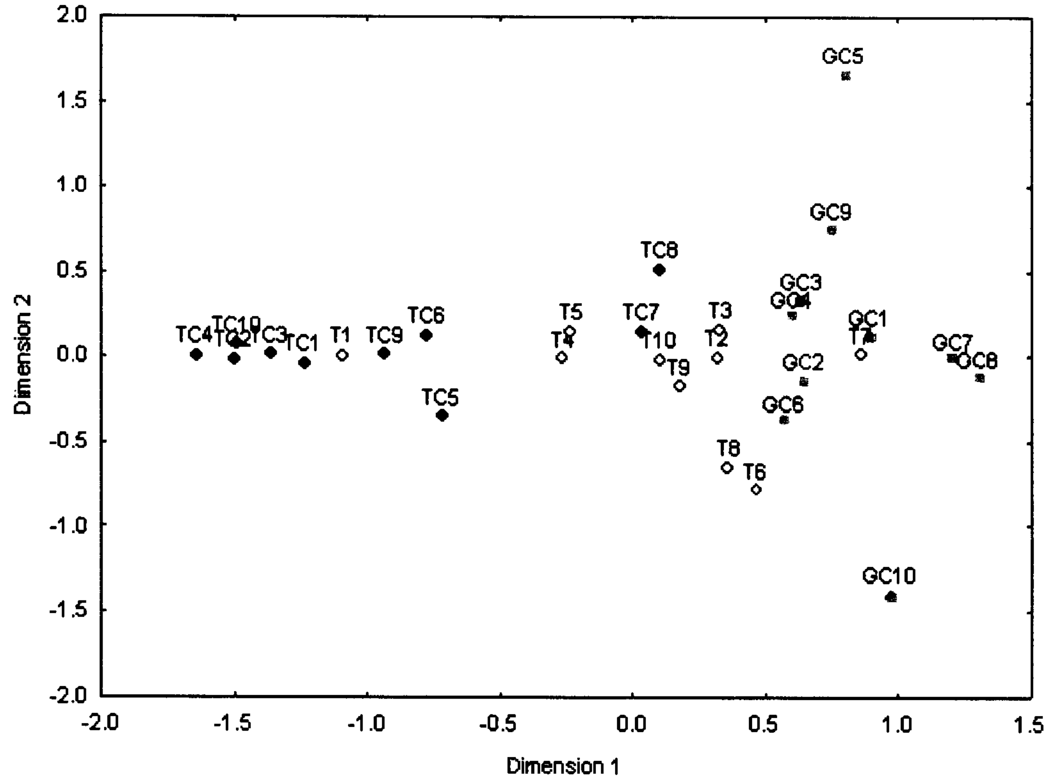

Figure 5.15. Multidimensional scaling of ground cover of all plots after two growing seasons. TC = tree-covered control plots, T = treated plots, and GC = grassland control plots.

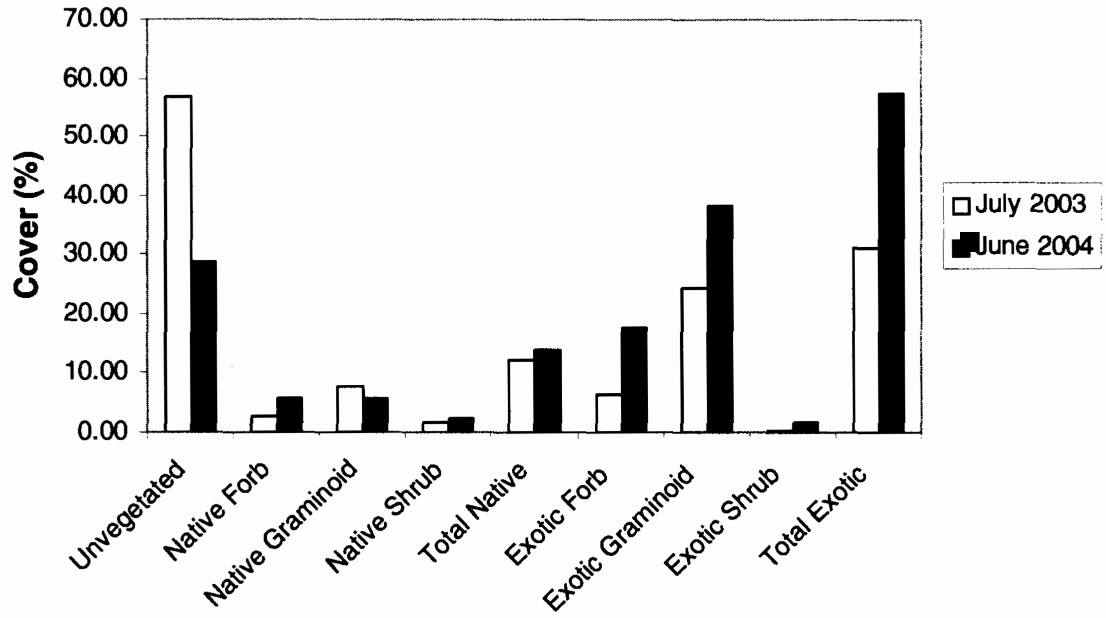

Figure 5.16. Ground cover of treated plots in July 2003 and June 2004, one and two growing seasons after treatment.

5.4. Discussion

Accounts of early settlers on Hornby Island indicate that the grasslands at Helliwell Provincial Park formerly covered a much larger area (Penn and Dunster 2001). In 1875, the area including what is now Helliwell Provincial Park was surveyed by J. Carey. The survey map delineates an area of "grassy hills" as distinct from the forest. A comparison of these grassy hills to the present distribution of grasslands reveals extensive forest encroachment (Figure 5.17). The surveyor's account suggests that forest encroachment was already taking place in 1875, as he notes the presence of young Douglas-firs in the grassland, and the small diameter of trees in the adjacent forest area.

Some of the area described as grassy hills in 1875 is today classed as older-growth forest (Penn and Dunster 2001).

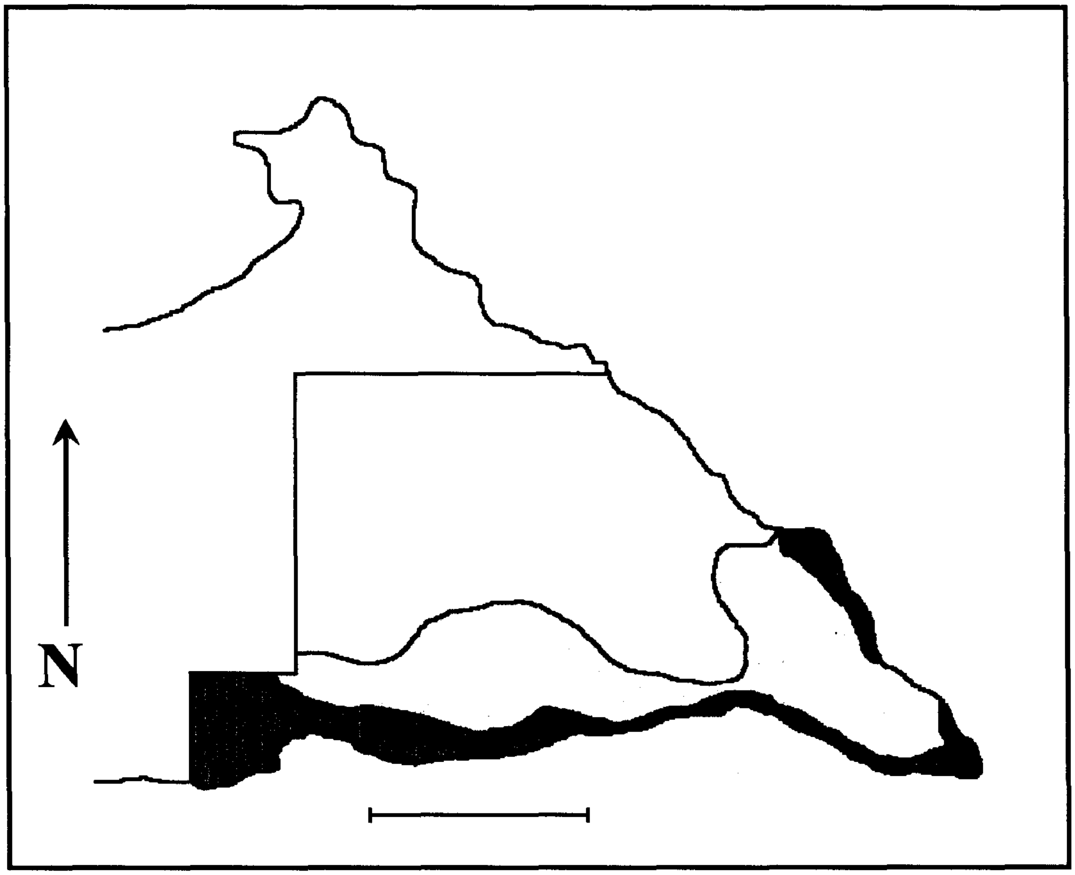

Figure 5.17. Change in area of grassland in Helliwell Provincial Park. Dark grey shading shows present distribution of grassland. Light grey shading shows the grassy hills of 1875. Scale bar represents 500 m. Modified from Penn and Dunster (2001).

Mapping of Douglas-firs showed that most trees growing in the grassland have a minimum age of between five and twenty years. If the actual age of the trees is estimated to be no more than five to ten years greater than the minimum age, the average age is approximately twenty years, and the oldest tree may be fifty-five. This is consistent with what is known of the history of Helliwell Provincial Park.

Before settlement, Helliwell Provincial Park, and other areas of Hornby Island, were used as food gathering areas by indigenous peoples (Penn and Dunster 2001). The management of root vegetables, such as camas, onions, and bracken fern, is believed to have included periodic burning of the harvest areas to maintain habitat for these plants. On Hornby Island, these practices would have ended by the mid to late 1800's, when the people who had used the island were largely eliminated by disease and war (Penn and Dunster 2001). As indicated by the land surveyor's notes, forest encroachment must have been rapid following the cessation of the traditional burning practises. Forest encroachment would then have been halted again in the early 1900's, when settlers began grazing sheep and cattle in the grasslands. Sheep and cattle grazing continued until the 1940's, with some grazing by feral sheep and goats thereafter (Penn and Dunster 2001). This grazing may account for the young ages of the trees growing in the grasslands. Studies of a similar ecosystem on Yellow Island, Washington, have suggested that encroachment of Douglas-fir trees into grasslands in the last twenty years may be due to the moderation of the summer climate by the few older trees (Agee and Dunnwiddie 1984). The established trees reduce the speed of desiccating winds, creating a more favourable environment for new seedlings. In Helliwell Provincial Park, encroaching trees frequently occur in groups. Those trees growing on the windward side of the group are often damaged, with multiple dead tops. Trees on the lee side of the group show a straighter growth form, with no wind damage. The lee side of the groups also often contains young trees (Figure 5.18).

Figure 5.18. Young trees growing on the lee side of established trees, and showing healthy growth form.

Recruitment of trees into the grassland does not appear to be episodic. Figure 5.4. shows that the majority of trees have become established in the last twenty years, but does not show multiple peaks, which would be expected if recruitment were episodic. The histogram shows very few trees less than five years old, but this may partly be an artefact of the difficulty in accurately aging such small trees.

A significant amount of grassland vegetation survived in the deep shade of the encroaching Douglas-firs and responded well to release when the trees were removed. Most of these were species that produce underground storage structures. Examination of control plots under standing trees showed that these species persist in a vegetative state, generally producing only a single prostrate leaf each year. However, the low density of these plants shows that eventually the underground reserves will run out and the plant will die of shading. Dunwiddie (2002) also removed young Douglas-firs from a grassland area. The trees were 10m tall and estimated at 15 years old. A small number of native forbs and grasses were found surviving under the shade of the trees. These were mostly lilies, but native grasses were also noted.

Non-native species also survive in the shade of encroaching Douglas-firs. This is shown by the lack of significant differences in the cover of exotic species between the tree-covered control plots and the grassland control plots. The environment created by tree removal also appears to favour non-native species. The major change in treated areas between 2003 and 2004 was the colonization of bare ground by exotic species. The cover of native species changed very little. In a Douglas-fir removal study in a similar ecosystem, the treated areas were rapidly colonized by exotic species, which then persisted: thirteen years after tree removal, the treated areas are dominated by exotic species (Dunwiddie 2002). Neither the native species that were already present, nor those growing in the adjacent grassland, have increased in the treated areas.

Multidimensional scaling has shown that, despite a general lack of significant differences in the cover of the various functional groups, the treated plots and the two types of control plots do differ in the species present and their abundances. It also shows that the treated plots are intermediate between the two types of control plots. This is important, as it shows that the disturbance caused by tree removal has not led to a new plant community, but one that has features of both the degraded plant community (i.e., the tree-covered control plots) and the desired future plant community (i.e., the grassland control plots).

Multidimensional scaling also provides insights on how future tree removal should proceed in the future. One treated plot (T1) clusters with the tree-covered control plots. T1 is located in the shade of established trees. The lack of light probably increased the mortality of grassland vegetation as the tree became established, leaving few grassland plants to be released by the treatment. In addition, the shading may retard the recovery of grassland vegetation in the treated area. Two tree-covered control plots (TC7 and TC8) cluster with the treated plots, suggesting that they are more similar to the grassland plots than the other tree-covered plots. TC7 and TC8 are the tree-covered control plots that cover the smallest area, and are both isolated plots, not bordered by other trees. These two factors mean that the area covered by the trees is not shaded as heavily as other tree-covered areas. This higher-light environment may increase the ability of grassland vegetation to persist under the trees. The treated plot that covers the smallest area (T7) clusters with the grassland control plots, suggesting that a sufficient amount of grassland vegetation has responded to release to approximate the desired plant community. If more trees are removed for the purposes of grassland restoration, removing smaller trees and trees that are not shaded by others will increase the success of the restoration.

Most studies that have attempted to restore open ecosystems through the removal of woody species have been conducted in eastern North America. These studies have been located primarily in oak parkland/ prairie ecosystems, and have usually included prescribed burning. These eastern North American projects have been very successful at recreating an open structure, while increasing the diversity and cover of native herbaceous species and reducing the prevalence of exotic species (White 1983, McCarty 1998, Abella *et al.* 2001). Experiments involving woody species removal and/ or burning in Garry oak and similar ecosystems have produced mixed results. Dunwiddie (2002) and MacDougall (2002) both experimented with fire in Garry oak and associated ecosystems, and both found response to fire to be species-specific and highly variable. Burning did not consistently favour native or exotic species, and individual species did not respond in the same way to each fire event. Studies of fire in the south Puget Sound have shown that burning at three to five year intervals increases native species and some exotic species, while longer or shorter intervals are detrimental to the native grassland (Tveten and Fonda 1999). In the Willamette Valley of Oregon, one- and two-time burns have no effect on the density of woody species (Pendergrass *et al.* 1998). These studies indicate that the use of fire as a restoration tool in Helliwell Provincial Park would be ill advised unless small-scale experiments can be done within the park first.

This study has shown that in some situations, native grassland that has been invaded by conifers can be restored through mechanical tree removal only. In Helliwell Provincial Park, tree removal is sufficient to restore grassland where the trees are small, or where the area covered by the trees is not too densely shaded. However, in the treated plots, two important components of the grassland flora are poorly represented. These are native graminoids and plants important to the life cycle of Taylor's checkerspot. These components would have to be added to treated areas if treatment were to continue on an operational scale. In addition, in areas where trees are large and light levels are very low, removal of trees is likely to result in areas of exotic flora that will not readily succeed to native grassland. In these areas, native species will also have to be added to restore native grassland. Studies in similar ecosystems have found that introducing plants as plugs is the most economical way to establish native species in a restoration site (Dunwiddie 2002, Vance *et al.* 2004).

If a sufficient area of native grassland, including mesic areas, could be restored, there would be no other known barrier to the success of a reintroduction of Taylor's checkerspot to Helliwell Provincial Park.

5.5. <u>References</u>

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6. Conclusions and recommendations

Southeastern Vancouver Island and the Gulf Islands feature many characteristics likely to lead to butterfly declines: large-scale habitat destruction, high number of species with high habitat specificity, long history of land management, and recent disruption of that management. If butterflies in general are to be conserved in this area, a proactive approach to habitat conservation and management must be applied on a landscape-scale. However, for species that have already severely declined or become extirpated, more specific action will be required. A historical perspective on butterfly conservation suggests that programs aimed at single species must be based on sound ecological information, must include provisions for ongoing habitat management, must include a high level of approval from the public and other stakeholders, and must be well-funded. It is my hope that this study and the recommendations that follow will form the embryo of such a program for the island large marble and Taylor's checkerspot.

6.1. *Euchloe ausonides insulanus*

The cause of the original extirpation of the island large marble will never be known with certainty. However, there is no evidence that this species could not survive in its former range in Canada, in areas of suitable habitat. The single extant population of *E. a. insulanus* thrives in a disturbed habitat dominated by exotic species. Similar habitat probably can be found or created within its former range in British Columbia. This species is a good candidate for a reintroduction attempt because of its apparently simple habitat requirements. Because only one extant population is known, and because all known former occurrences are in Canada, reintroduction is a logical step in the conservation of this narrowly endemic subspecies. The following activities are recommended for the conservation of this species:

1. Continue to search for acceptable habitat

Habitat similar to that of the single extant population has not yet been found in BC. Disturbed, weedy areas within the former range of the butterfly should be investigated to determine whether populations of the known host plants are present, and whether they would be suitable as reintroduction sites. Coastal areas should also be investigated to determine whether there are sites with an abundance of the only known native host plant that would be suitable as reintroduction sites.

2. Begin the creation of new habitat

Euchloe ausonides insulanus is not likely to benefit from general ecosystem restoration, as its habitat requirements can be easily met by disturbed areas, and the only known native habitats are quite specialized. Instead, areas should be located that have low conservation value, such as abandoned agricultural lands, roadsides, or developed parks. At these sites, certain areas should be designated for production of host plants. These areas should be seeded with the known host plants, at a high seeding rate. The soil will need to be periodically disturbed to ensure continuity of habitat for the host plants. This disturbance should be done in limited areas on a rotational basis, in order to minimize disturbance of a reintroduced butterfly population.

3. Reintroduce butterfly

Once suitable habitat has been located or created, *Euchloe ausonides insulanus* should be reintroduced from the San Juan Island population. Which life stage to reintroduce, how many individuals to reintroduce, and whether or not to include captive rearing should be determined through literature review and discussion with American conservation authorities prior to any reintroduction attempt.

6.2. ***Euphydryas editha taylori***

Studies of Taylor's checkerspot have shown that the quality of existing habitat is low, and that this may explain the extirpation of this species from Canada. However, these studies have not found any reason that this butterfly would not survive in Canada if the quality of habitat could be improved. While this species has declined severely throughout its range, British Columbia is the only major jurisdiction from which it has been extirpated. British Columbia is therefore a likely place to begin any reintroduction program, and the reestablishment of this species in British Columbia would be an important step in improving the animal's conservation status. The following activities are recommended for the conservation of this species:

1. Continue the search for acceptable habitat and extant populations

This study has shown that the ecological range of *Euphydryas editha taylori* is greater than previously known. Populations occur at higher elevations than previously known, and utilize disturbed areas to some extent. Surveying for extant populations should continue with this new information. At the same time, the search should continue for areas of mesic meadow habitat that would be suitable as reintroduction sites.

2. Restore habitat at sites of historic populations

Important features of habitat may still include some unknowns. In order to minimize the probability that these unknowns will cause a reintroduction attempt to fail, reintroduction should begin at formerly occupied sites. However, before this can happen, restoration will have to be done to improve quality of habitat. At most potential sites, woody encroachment is an important issue that must be addressed prior to reintroduction. The restoration of mesic areas is critical to success. When removal of woody vegetation is likely to result in bare ground that will be colonized by non-native plants, plugs of native plants should be added as the woody vegetation is removed. When removal of woody vegetation will not result in an increase in the host plants of *E. e. taylori*, these plants (particularly native species, as non-natives are likely to colonize) should also be introduced as woody vegetation is removed. General ecosystem restoration that includes maintenance of open meadows is likely to improve potential habitat for *E. e. taylori* and increase likelihood of a successful reintroduction.

3. Reintroduce butterfly

Once mesic habitat has been restored or new habitat has been located, *Euphydryas editha taylori* should be reintroduced from the United States. The origin of the individuals used, which life stage to reintroduce, how many individuals to reintroduce, and whether or not to include captive rearing should be determined through literature review and discussion with American conservation authorities prior to any reintroduction attempt.