

AN ABSTRACT OF THE THESIS OF

Richard P. Young for the degree of Doctor of Philosophy in

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Title: Fire Ecology and Management in Plant Communities of  
Malheur National Wildlife Refuge Southeastern Oregon

Signature redacted for privacy.

Abstract approved: \_\_\_\_\_

Richard F. Miller

This research assesses prescribed burning as a habitat management technique in wetlands and associated upland communities of Malheur National Wildlife Refuge, southeastern Oregon. Experimental burns were conducted to evaluate fire behavior and effects in wetland and upland habitats, and, fire effects on Cirsium arvense. Wetland plant communities were monotypic stands of emergent macrophytes: Scirpus acutus, Sparganium eurycarpum, Carex atherodes, Juncus balticus, Eleocharis palustris. Upland communities studied included Chrysothamnus nauseosus/Elymus cinereus and Sarcobatus vermiculatus/Distichlis spicata shrub-grasslands and an Elymus triticoides mesic meadow. Burning was conducted during periods of vegetation dormancy.

Stepwise multiple regression was used to evaluate the relative influence of various fuel and weather parameters on fire behavior, and, to quantify the relationship of the "best" independent variables to the response variables rate-of-spread (ROS) and flame length. Models incorporating windspeed alone, or in combination with a second variable, account for 50 to 90 percent of the variation in ROS. Successful burns were conducted with a wide range of conditions, provided the surface of fuels were dry and winds were steady. Burning prescriptions and techniques suitable for these fuel types are given.

Fire effects on vegetation were measured for two years (Sarcobatus/Distichlis, one year). Vegetation response was largely insensitive to timing of burns within the dormant period. Comparisons with unburned treatments (=non-use) indicate fire significantly alters vegetation structure and community function; however, responses were often species-specific. Aboveground herbage production increased for one to two years in all but winter-burned Sparganium communities. Burning increased shoot density of rhizomatous species. Vigorous postburn sprouting-regrowth of shrubs resulted in rapid replacement of canopy cover and volume. Reproductive effort varied markedly among species and in response to fire. The observed dynamics of organic residues suggest these communities will return to preburn status after three to five years.

Dormant season burning reduced relative abundance of Cirsium arvense. Changes in population structure and reproductive success, as

well as increased production of associated species, indicate burning may be a useful means of halting Cirsium invasion or spread by maintaining a productive stand of native vegetation.

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## PREFACE

### Thesis Organization and Format

This thesis is composed of five sections. Specific results of the research project are reported in Sections II-V which partition the information into logical units. These include sections on (1) fire behavior and burning prescriptions, (2) fire effects on wetland and (3) adjacent upland vegetation, and (4) response of the noxious weed Canada thistle to burning. These sections are preceded by an introductory chapter which includes a review of pertinent literature and a description of the study area. A final section summarizes the findings of this project and considers implications of fire as a management tool of inland wetland/uplands in general, and wildlife habitats of Malheur Refuge in particular.

The thesis utilizes a manuscript format as it is anticipated Sections II-V will be submitted for publication as separate papers. Within the context of thesis preparation requirements specified by the Oregon State University Graduate School, this thesis follows the format designated by the Journal of Applied Ecology, Blackwell Scientific Publications, London. Numbering of figures and tables is continuous within individual sections. For example, the first figure of Section I is labelled Figure I.1, the first figure of Section II is labelled Figure II.1, and so on. Citations of published literature are given in a common list of References at the end of the thesis.

## Contribution of Authors

Richard F. Miller served as major advisor and co-investigator during all phases of this research program. Robert E. Martin was a co-investigator in the study of fire behavior, providing assistance in conducting and monitoring the controlled burns and in formulating the burning prescriptions. Richard F. Miller is co-author to sections II, III, IV and V; Robert E. Martin is second co-author to section II.



SECTION I

FIRE ECOLOGY AND MANAGEMENT IN PLANT COMMUNITIES OF  
MALHEUR NATIONAL WILDLIFE REFUGE  
SOUTHEASTERN OREGON

FIRE ECOLOGY AND MANAGEMENT IN PLANT COMMUNITIES OF  
MALHEUR NATIONAL WILDLIFE REFUGE, SOUTHEASTERN OREGON

INTRODUCTION

Fire is considered to be a natural component of wetland ecosystems (Lynch 1941; Komarek 1962, 1964; Vogl 1969; Davis 1979). Furthermore, it has been proven to be an important wildlife management tool in wetlands of North America (Bendell 1974; Wright & Bailey 1982; Kirby & Lewis 1983). Despite this, detailed knowledge of fire and its effects upon wildlife species and habitat derives largely from research and management efforts in terrestrial ecosystems. The bulk of published information on fire effects and management in North American wetlands concerns vegetation and wildlife in Florida (Robertson 1962; Klukas 1973; Vogl 1973; Bancroft 1976; Wade et al. 1980) and Gulf Coast marshes (Lynch 1941; Singleton 1951; Givens 1962; Hoffpauer 1968; Perkins 1968; Oefinger & Scifres 1977). Whereas some studies have dealt with wetland ecosystems of the midwestern United States (Schlichtemeier 1967; Vogl 1964, 1967) and Canada (Ward 1968), little is known about controlled burning and fire effects on inland marshes of western North America (Smith and Kadlec 1985).

The beneficial use of fire for habitat management is dependent on understanding fire behavior within and fire effects on a specific type of ecosystem (Wright & Bailey 1982). Recognizing the potential uses and lack of information concerning fire as a wildlife habitat management tool in western inland wetlands, a cooperative project

involving the Eastern Oregon Agricultural Research Center (Burns), Oregon State University, and the US Fish & Wildlife Service was initiated in 1980. The purpose of this study was to evaluate the use and effects of controlled burning in wetland habitats of Malheur National Wildlife Refuge (NWR) in southeastern Oregon. This paper reports results obtained from this project.

### Objectives

This project includes aspects of both fire ecology and fire management at Malheur NWR. Study objectives may be summarized as follows:

- (1) To evaluate the effects of prescribed fire on major wetland and upland plant communities.
- (2) To compare results of burns conducted at various times during the fall-early spring period of vegetation dormancy.
- (3) To evaluate the effects of various fuel and weather parameters on fire behavior.
- (4) To develop burning prescriptions and techniques suitable for wetland vegetation.

### REVIEW OF LITERATURE

#### Fire as a Component of Wetland Ecosystems

Wetlands typically are areas of high net primary production (NPP) (Leith & Whittaker 1975). Decomposition of plant residues, however,

often lags behind production, resulting in a net accumulation of organic materials (Leisman 1953; Reader & Stewart 1972). This imbalance has been shown to be responsible for the progression from one community type to another in wetlands (Pearsall 1920; Wilson 1935); and in fact, the accumulation of organic and/or inorganic sediments is frequently cited as the driving force in hydrarch succession from wetlands to terrestrial ecosystems (Valk 1981). Palynological studies, however, have repeatedly shown wetland vegetation to be more stable over geological time than associated terrestrial communities (Watts & Winter 1966; Watts & Bright 1968). Fire can have a significant role in the maintenance of wetland vegetation, and thus, as in terrestrial communities, is an important factor in the structure and functioning of wetland ecosystems (White 1979; Pyne 1982).

Lynch (1941) considers fire to have always been a factor in the ecology of Gulf Coast marshes. Large accumulations of organic debris coupled with frequent late summer drought often lead to lightning-caused fires. Fires initiated by lightning have been observed to last for days, even in wetlands regularly burned (Lynch 1941).

Additional evidence substantiating the historical importance of fire in wetlands comes from analysis of pollen and charcoal records in organic deposits. Davis (1979) used this approach to demonstrate the susceptibility of wetlands to fire at Tamarack Creek in northern Wisconsin. High fire frequency, indicated by charcoal concentrations, was associated with emergent aquatic and sedge meadow communities;

whereas, decreasing frequency of charcoal was positively correlated with later stages of succession, including shrub-carr and bog shrub communities. Vogl (1969) similarly showed periodic fire produced relative stability in a sedge-peat wetland of north-central Wisconsin. Open marsh, sedge meadow, and wet prairie communities were held in dynamic equilibrium by alternations of wet period flooding and fires during drought. Even before the presence of man, fires initiated by lightning have acted as an evolutionary force in a wide variety of ecosystems (Komarek 1962, 1964). Natural and man-caused fires have certainly been an integral part of the environment both regionally and within communities of Malheur NWR (Griffiths 1902, 1903; Shinn 1977). Lightning fires have burned several thousand acres of terrestrial and wetland communities on the refuge every summer during the course of this study (personal observation and unpublished reports on file at Malheur NWR, US Fish & Wildlife Service, 1980, 1981, 1982). These fires have started both in wetland communities and in adjacent uplands, subsequently spreading into wetland areas. Annual narrative reports for Malheur NWR indicate such fires have occurred in most of the years for which records have been maintained. Fire is undoubtedly an important component of wetland ecosystems at Malheur NWR.

#### Effects of Fire on Wetland Vegetation

Wetland fires fall into two categories (Lynch 1941; Hoffpauer 1968). The most common fire is a surface or cover burn. Surface fires remove aboveground or above-water stems and leaves of plants.

These fires occur when water levels are at or above the root horizons, when water or soils are frozen and/or when air temperatures are low. In contrast, root, peat, or ground fires oxidize part or all of the root mat and accumulated organic deposits. Such fires generally occur as wildfires during dry periods, especially during times of extended drought.

Vogl (unpublished report submitted to the US Fish & Wildlife Service, 1980) summarized the effects of fire in wetland ecosystems. His comments are based on published information, personal experience, and communications with land managers using fire (but not recording or publishing their results) for wetlands management. The resulting generalizations incorporate information from a variety of wetland ecosystems and are founded on the premise that wetlands have more properties in common, than differences, with respect to fire effects.

#### Fire differentially favors herbaceous over woody species

As phanerophytes, large portions or all aboveground regenerative (meristematic) tissue of woody plants is subject to heat injury by fire. Species capable of resprouting from undamaged meristems near or below the soil surface must replace aboveground biomass lost to fire in order to regain preburn levels of abundance. Most herbaceous species, however, are cryptophytes and hemicryptophytes. Surface burns generally remove only the aerial portions of these plants (which have typically died back prior to the periods when most fires occur) and have little or no effect on meristematic tissues which initiate regrowth (Volland & Dell 1981). Therefore, although woody species

may not be eliminated by fire, they are usually reduced in density, relative dominance, and/or rate of spread. Differential favoring of herbaceous species is strengthened by frequent burning (annually or every few years).

Vogl (1964) has documented this principle following prescribed burning in muskegs of northern Wisconsin. Results indicate fire produces a retrogression from conifer swamp dominated by trees to sedge and ericaceous shrub dominated bog or muskeg. These communities may be further altered to sedge meadows which support a minimum of woody vegetation with repeated burning. Similarly, Wade et al. (1980) have observed herbaceous species increase at the expense of woody plants with increasing fire frequency in a variety of southern Florida ecosystems.

Fire differentially promotes plant species that reproduce vegetatively

Rhizomatous graminoids and grass-like monocots constitute the bulk of live standing crop phytomass in wetlands dominated by herbaceous species (Cruz 1978). The herbaceous component has been shown to retain or increase relative dominance following surface burns in communities of Phragmites communis L. (reed grass) in Nebraska (Schlichtemeier 1967) and Manitoba (Ward 1968), Cladium jamaicense (Wats.) O'Neil (sawgrass) in southern Florida (Wade et al. 1980), Scolochloa festuacea (Willd.) Link. (white top grass) in Manitoba (Ward 1968), and Spartina spartinae (Trin.) Hitch. (Gulf cordgrass) in southern Texas (Oefinger & Scifres 1977). With respect to woody species, Vogl (1964), in his study of fire effects on northern

Wisconsin wetlands found the only postburn increasers were Betula pumila L. (bog birch) and several species of Salix (willows), all strong resprouters.

#### Fire stimulates plant productivity

Increased productivity is reflected in increased growth rates, size of plant parts, production of floral components, seed production, and aboveground herbaceous production. Increased production frequently occurs as a pulse for one to three years after a fire, thereafter diminishing rapidly.

In the first year after burning portions of an inland Florida pond Vogl (1969) found aerial standing crop of burned areas increased over 50% in aquatic emergent vegetation and greater than 250% in wet mesic vegetation, relative to unburned areas. Burning produced more rapid growth, and plants of wet mesic sites were ultimately "taller and more robust" and displayed higher stem densities. Additionally, increased flowering activity of the aquatic emergent Polygonum hydropiperoides Michx. (water pepper) was observed on the burned site. In earlier burning studies in northern Wisconsin Vogl (1964) found fruit and seed production of several graminoids and shrub species were greatly increased in the year after burning. Oefinger and Scifres (1977) noted that prescribed burning of Spartina spartinae increased both growth rates and total NAPP at three sites on the Texas Gulf Coast. Comparing herbage production on flood meadow pastures in southeastern Oregon, Britton et al. (1980) found burn plots produced  $810 \text{ g m}^{-2}$ , where non-use plots produced  $544 \text{ g m}^{-2}$  in the year after



treatment. This was attributable in part to the greater average height growth of vegetation on burn plots (122 cm vs. 69 cm).

In contrast to surface burns, ground fires remove part or all of the root mat and/or organic deposits, and thus can reduce community NAPP, may result in drastic changes in species composition, and may even produce open water within stands of emergent vegetation. In Gulf Coast marshes Lynch (1941) states root burns can remove dense stands of climax Cladium jamaicense. This allows early successional plants such as Eleocharis sp. (spikerush), Catalia sp. (waterlily), and Echinochloa sp. (wild millet) to become established. In drought years deep peat burns can produce open pools where Catalia sp., Brasenia sp. (water shield), and Ruppia sp. (widgeon grass) invade. Similar results have been observed in the Delta Marsh of Manitoba (Ward 1968) where peat substrates, ignited by lightning in dry years, are completely consumed and open pools were produced. Burning when root mats are only partially dried, however, results in less drastic changes. Production, height, and density of Phragmites communis and Scolochloa festuacea communities were reduced for two years when prescribed burning was conducted with these conditions (Ward 1968). This allowed Chenopodium rubrum L. (red goosefoot), a valuable waterfowl food, to invade and set seed in both community types.

#### Uses of Fire in Wetlands: Wildlife Management

Much of wildlife management efforts involve manipulation of habitat to improve food and cover conditions (Dasmann 1964). Fire,

through its effects on vegetation, can therefore be a useful tool in managing wildlife populations of wetland habitats. Often, multiple objectives may be realized from a single burn. Winter burning in marshlands of the Nebraska Sandhills (Schlichtemeier 1967) reduced residual stem density of Phragmites communis and Scirpus sp. (bulrush) by 85% and 60%, respectively. Observations from this and other burns indicate fire produces more open water, greater edge between open water and emergent vegetation, and opens otherwise impenetratable vegetation for waterfowl feeding and resting. In addition, the abundance of desirable waterfowl food plants both in open water and among emergent vegetation are increased. Similar results have been obtained in Delta Marsh of Manitoba (Ward 1968) where spring burns are used to thin dense stands of Phragmites communis and Scolochloa festuacea, increase production of the waterfowl food species Chenopodium rubrum, check encroachment of woody species, and create more water-vegetation edge for nesting waterfowl.

Fire is used in Gulf Coast wildlife refuges to improve production of and/or access to wildlife plant foods (Lynch 1941; Givens 1962; Perkins 1968). Periodic surface burns are essential for management of geese and muskrat. These burns remove cover and open stands of Carex sp. (sedges) and grasses, providing access to roots and rhizomes for foraging geese. Scirpus olneyi Gray (Olney bulrush), a key management species for muskrat, must be burned at least every other year to maintain open, productive stands and to prevent succession to less desirable plant species.

A number of North American waterfowl species nest in the mesic herbaceous uplands adjacent to wetland communities. Several papers emphasize the value of dense, undisturbed stands of this vegetation in promoting nesting and hatching success (Deubbert 1969; Schrank 1972; Deubbert & Lakemoen 1976). Excessive buildup of litter, however, eventually reduces the productivity of these sites (Ehrenreich 1959; Hulbert 1969) and can result in encroachment or increased cover of undesirable woody species (Vogl 1965; Ward 1968). Such conditions lower the value of these communities for nesting waterfowl. Thus, the management practice of non-use, which produces dense, undisturbed stands of herbaceous vegetation, is ultimately self-destructive. Periodic use of fire, however, initiates a new cycle of improved nesting habitat by removing accumulations of litter and checking the advance of woody plants (Vogl 1967; Kirsch & Kruse 1973; Bellrose & Low 1978).

#### Fire Effects on Upland Vegetation

As previously indicated, upland communities adjacent to wetlands provide valuable habitat for certain waterfowl species as well as a variety of other birds and mammals (Vogl 1967). Furthermore, it is often not possible nor desirable to manage uplands and wetlands separately due to their mosaic pattern of occurrence and integration as a complex ecosystem. Therefore, any use of fire as a management tool in wetlands must consider the response of the associated terrestrial vegetation.

Upland plant communities of Malheur NWR are included within the Artemisia tridentata Nutt. (big sagebrush) and Atriplex confertifolia (Torr. & Frem) Wats. (shadscale) zones described by Holmgren (1972). Although the specific communities dealt with in this study have not been previously considered in fire ecology research, abundant data are available concerning the response to fire of many plant species in this major zone (see reviews by Wright et al. 1979 and Wright & Bailey 1982). A list of these plants and their response to fire is given in Table I.1.

#### Conditions for Burning in Wetlands

Recommendations for prescribed burning should include the following characteristics (Martin & Dell 1978; Wright & Bailey 1982):

- (1) They should be specific to a fuel type; that is, a certain class of vegetation for which, other variables held constant, fire behavior will be roughly similar for different burns. The fuel type defines the range of fuel loading (i.e. weight of fuels per unit area) of different fuel size classes, and to some degree, the structural arrangement of fuels, including continuity, height and/or cover.
- (2) They should describe the weather conditions in which safe burning may be conducted. This will include the acceptable range of air temperature, relative humidity, and wind speed.

Table I.1. Summary of fire effects on plant species occurring in study areas of the Elymus triticoides, Chrysothamnus nauseosus/Elymus cinereus; and Sarcobatus vermiculatus/Distichlis spicata upland communities, Malheur NWR; ---: no data available.

| Growth habit   | Response to fire               | Recovery time (yr) | Comments  | References   |
|--|--------------------------------|--------------------|---|--|
| Species  |                                |                    |   |  |
| Shrubs   |                                |                    |   |  |
| <u>Artemisia tridentata</u>                                | Severely damaged               | 30                 | Nonsprouter; reestablishment by seed  | Pechanec & Stewart 1944; Blaisdell 1953  |
| <u>Chrysothamnus nauseosus</u> and <u>C. viscidiflorus</u> | Enhanced                       | 10-25              | Vigorous sprouters; often produce heavy seed crops after fire   | Blaisdell 1953   |
| <u>Sarcobatus vermiculatus</u>                             | Undamaged                      | ---                | Vigorous sprouter   | Daubenmire 1970  |
| Grasses & Grasslikes                                       |                                |                    |   |  |
| <u>Bromus tectorum</u>                                     | Slightly damaged<br>- enhanced | 1-3                | Annual; seed reserves severely depleted, but plants are vigorous and produce abundant seed for 1-2 years after fire | Pickford 1932; McKell 1950; Young et al. 1976                                  |
| <u>Carex douglasii</u>                                     | Undamaged                      | 1-3                | Rhizomatous; mod-severe damage with spring burns  | Pechanec & Stewart 1944; Countryman & Cornelius 1957<br>Anderson & Bailey 1980 |
| <u>Distichlis spicata</u>                                  | Undamaged                      | 1                  | Rhizomatous; burning & flooding severely reduces production   | Hadley 1970; Smith & Kadlec 1985   |
| <u>Hordeum brachyantherum</u>                              | Moderately damaged             | ---                | Response may have been due to climate or climate x burn interaction   |  |
| <u>Hordeum jubatum</u>                                     | Slightly-severely damaged      | ---                | Bunchgrass; greatest damage with spring burns   | Hadley 1970  |
| <u>Poa nevadensis</u>                                      | Slightly-damaged               | 1-3                | Bunchgrass; minimal damage with late summer & fall burns  | Harniss & Murray 1973  |
| <u>Poa pratensis</u>                                       | Slightly-severely damaged      | ---                | Rhizomatous sodgrass; greatest damage with spring burns   | McKell 1950; Hadley 1970   |

Table I.1 -- Continued

Forbs, perennial

|                               |                                 |     |   |  |
|-------------------------------|---------------------------------|-----|---|--|
| <u>Achillea millefolium</u>   | Slightly damaged<br>- enhanced  | 5   | Rhizomatous; may increase considerably<br>when fire reduces competition | McKell 1950; Blaisdell 1953;<br>Countryman & Cornelius 1957;<br>Anderson & Bailey 1980 |
| <u>Agoseris glauca</u>        | Slightly damaged                | --- | Taprooted   | Anderson & Bailey 1980   |
| <u>Artemisia ludoviciana</u>  | Undamaged<br>- enhanced         | --- | Rhizomatous   | Dix 1960; Kirsch & Kruse 1972;<br>Anderson & Bailey 1979, 1980                         |
| <u>Cirsium arvense</u>        | Slightly damaged<br>- enhanced  | --- | Rhizomatous   | McKell 1950; Dyrness 1973  |
| <u>Cirsium vulgare</u>        | Slightly damaged<br>- enhanced  | --- | Short-lived perennial   | Dyrness 1973   |
| <u>Potentilla gracilis</u>    | Slightly damaged<br>- undamaged | --- |   | Nimir & Payne 1978   |
| <u>Solidago canadensis</u>    | Undamaged<br>- enhanced         | --- | Rhizomatous   | McKell 1950  |
| <u>Solidago missouriensis</u> | Undamaged<br>- enhanced         | --- | Rhizomatous   | Gartner & Thompson 1972;<br>Wolfe 1972; Anderson &<br>Bailey 1980                      |
| <u>Vicia americana</u>        | Undamaged<br>- enhanced         | --- | Rhizomatous   | McKell 1950; Anderson &<br>Bailey 1979, 1980   |

Forbs, annual: all species considered here are reportedly enhanced to slightly damaged by fire;  
all may be abundant in early stages after fire; many are susceptible to spring  
burns; includes the following species --

McKell 1950; Blaisdell 1953;  
Dix 1960; Daubenmire 1970;  
Harniss & Murray 1973;  
Barney & Frischknecht 1974

Alyssum alyssoides, Amsinkia tessellata, Chenopodium album, Collomia linearis,  
Cordylanthus ramosus, Descurainia pinnata, Gayophytum diffusum, Lactuca pulchella,  
Lappula redowskii, Polygonum douglasii, Salsola iberica, Sisymbrium altissimum

- (3) Both the season and time of day should be specified. The season of year relates strongly to fuel conditions and, in wetlands, the activities of wildlife species. Diurnal changes in weather help achieve prescription conditions.
- (4) In wetlands it is important to define soil-peat moisture conditions, or where applicable, water level or presence of ice.
- (5) Firing or ignition methods can be used to modify fire behavior under various conditions. The effect of firing technique on the resulting fire behavior is important in allowing variation in other variables while meeting management objectives.
- (6) After the above variables are set, the expected fire behavior should be predictable. In particular, the prescription should indicate flame length of the fire front and the potential for spotting; that is, the movement of airborne firebrands beyond the flame front. This information is critical for the safe use of natural and man-made firelines.

Burning prescriptions vary from place to place due to differences in local conditions. Therefore, most prescriptions are simply guidelines that must be tailored, through experience, to a specific area. As with fire effects on wetland vegetation, however, there are likely a number of principles common to the controlled use of fire in wetlands of different areas. It is instructive, therefore, to review available information concerning burning conditions in wetland communities. Unfortunately, few controlled burns have been

adequately documented, and even fewer authors have suggested specific recommendations for prescribed burning.

Fire is used extensively in Gulf Coast marshes for maintenance and improvement of wildlife habitat. Lynch (1941) suggests surface burns be conducted in alternate years to remove dense cover. Best results are achieved when sites have 6-13 cm of standing water and burned with either steady winds or at night with diminishing winds. Spotty burns result from dying or variable winds or in damp weather.

Schlichtemeier (1967) reported weather conditions during four days of controlled burning in marshes of the Nebraska Sandhills (Table I.2). The objective of the burns was to improve waterfowl habitat by removing old, dense stands of Phragmites communis and Scirpus sp. All burns were conducted in winter when marshes were frozen and the surrounding rangelands were covered by several centimeters of snow. This resulted in some unburned patches and strips; but in burned areas residual plant materials were consumed to a 5-15 cm stubble height over the snow. Densities of Phragmites and Scirpus were reduced 85% and 60%, respectively. Schlichtemeier concluded that despite considerable variation in relative humidity and temperature over the four days of burning, little variation in vegetation "combustibility" was observed.

Burning is utilized at Delta Marsh in south-central Manitoba for waterfowl habitat improvement (Ward 1968). Spring burning is employed to create greater nesting edge, to thin dense stands of marsh vegetation, and to remove invading woody species. Due to moist



Table I.2. Weather conditions during four winter burns in marshes of the Nebraska Sandhills (Schlichtemeier 1967).

| Date of burn     | Relative<br>humidity<br>(%) | Temperature<br>(C) | Wind Speed<br>(km h <sup>-1</sup> ) |
|------------------|-----------------------------|--------------------|-------------------------------------|
| 23 February 1966 | 72                          | -1                 | 24                                  |
| 24 February 1966 | 60                          | +3                 | 11                                  |
| 18 January 1967  | 67                          | -8                 | 37                                  |
| 19 January 1967  | 52                          | +6                 | 27                                  |

substrates these burns remove accumulated graminaceous vegetation but do not affect regrowth. Firing is conducted on bright, warm days after several preceding days of similar weather, and with wind speeds less than  $32 \text{ km h}^{-1}$ . Edges of bays and creeks and roads have been used as natural firelines. Mechanically cut and cleaned firelines of 6-9 m width are used by backfiring away from the lanes before igniting headfires. In some areas, formerly productive stands with even mixtures of Phragmites communis and Scolochloa festuacea have been replaced by dense and less useful Phragmites thickets. A summer burn was used to thin such a stand. By burning when the substrate was drying, the root mat was partially destroyed. This burn was conducted in July with an air temperature of  $32 \text{ C}$  and winds less than  $24 \text{ km h}^{-1}$ . Two years later Phragmites plants had not yet regained preburn levels of density or height. Growth of Scolochloa was favored by the burn and Chenopodium rubrum, a valuable duck food, had become established within the stand.

Vogl (1973) conducted a shoreline burn at a large northern Florida pond to investigate bird utilization and vegetation response in burned and unburned areas. Weather conditions at the time of the burn were clear with  $16 \text{ C}$  air temperature and moderate-gusty winds. Dry weather had persisted in the previous week. Ignition was accomplished by using a drip torch to light a line of fire along the water's edge. This produced a headfire with 2-4 m flames in the wet mesic vegetation. The fire also backed into the wind through emergent vegetation until reaching open water. In the areas of greatest fire

intensity vegetation was consumed to ground-level, however, basal portions of the larger herbaceous stems remained in most areas. In wet spots and places where vegetation had been heavily matted only the finer stems and leaves were consumed. Woody stems were still standing after the burn.

A preliminary study of prescribed burning was conducted in wet meadow vegetation at Malheur NWR (Britton et al. 1980). Preburn fuel load was  $7100 \text{ kg ha}^{-1}$  of continuous fine fuels. The burn was conducted in November of 1978. The results suggest prescribed fires in this vegetation type be conducted with air temperatures of  $10-18 \text{ C}$ , relative humidity of  $30-40\%$ , and  $0-3 \text{ km h}^{-1}$  winds (Wright & Bailey 1982).

Smith and Kadlec (1985) report weather conditions during controlled burning of a Great Salt Lake marsh in early September, 1981: mean wind speed= $16.6 \text{ km h}^{-1}$ , mean dew point= $5 \text{ C}$ , maximum temperature= $28.5 \text{ C}$ . Unfortunately, these authors provide no quantitative information on fuels or fire behavior for this burn.

It is apparent that prescribed burning in wetlands can be done in a variety of conditions, depending upon the area to be burned and the objectives. Vogl (unpublished report presented to the US Fish & Wildlife Service, 1980) states that most surface fires are rapid moving. Control is relatively easy, however, and is usually accomplished with a small number of personnel. A variety of natural and constructed firelines are suitable, especially when backfires are employed to widen lanes prior to head firing.

Few attempts have been made to conduct controlled peat fires as they tend to be slow moving and persistent, resulting in personnel commitment for long periods. Control is dependent on the ability to reflood an area once objectives have been met, or by trenching the perimeter to the water table or mineral soil (Vogl, previous citation).

### Cirsium arvense

Cirsium arvense (L.) Scop. (Canada thistle) is an aggressive perennial weed that reproduces from seed and rhizomes. Introduced from Europe, it is now widespread throughout much of the United States and Canada. Cirsium is well adapted to a variety of habitats and is responsible for substantial reductions in crop yields and both the quality and quantity of forages (Hodgson 1968).

Control measures have been developed primarily for cropland and improved pastures where economic returns are likely to be greatest. Early control efforts centered on cultural practices such as mowing, tillage, and competitive crops; but current control methods emphasize foliar applied herbicides (Hodgson 1958; Peterson & Parochetti 1978). These techniques, however, may not always be economically efficient or even possible to implement where Cirsium has invaded native rangelands. Comparatively low forage resource values, rough topography, and/or the large areas often involved, frequently are responsible for not utilizing proven agronomic practices on rangelands (Vallentine 1971). Furthermore, such measures may negatively impact wildlife populations within the treated areas. For example, herbicide

use may be restricted due to losses of nontarget broadleaf dicots (Blaisdell & Mueggler 1956; Thilenius et al. 1974). Often a desirable component of wildland ecosystems, forbs can be significantly reduced with applications of certain herbicides, resulting in decreased diversity of both the plant community (Tomkins & Grant 1977) and wildlife populations using these habitats (House et al. 1967).

Cirsium invasion of native rangelands appears to be a problem especially of highly productive, mesic habitats. Stachon & Zimdahl (1980) found high densities of Cirsium occurring in undisturbed, mesic herbaceous vegetation of northeastern Colorado. Cirsium reduces production of subirrigated Poa pratensis L. (Kentucky bluegrass) meadows of Nebraska by as much as 300% (Reece & Wilson 1983). In The Netherlands Cirsium is a serious problem in lowlands seeded to Phragmites australis (Cav.) Trin. ex Steudel (common reed) (Hemminga & Toorn 1970 -- cited in Toorn & Mook 1982). The following sequence has been described on these sites. About four to five years after Phragmites is established a closed vegetation develops, at which time Cirsium begins to invade. This development can be prevented, however, by burning the old stems and litter in the winter.

Extensive infestations of Cirsium occur along roadsides, canals, and sloughs of Malheur NWR in southeastern Oregon. These populations appear to be spreading into adjacent mesic herbaceous communities, and in fact, have already invaded and dominate significant portions of many fields. Refuge practices instituted over the last 10 to 15 years may, at least in part, be attributable for this situation. Delayed

haying dates have been set to allow a greater percentage of young birds to move off nests and into nearby marshes before cutting commences; thus reducing mortality of young birds. Additionally, some fields have been set aside for non-use with respect to haying and/or livestock grazing. Non-use allows vegetation to develop a dense, undisturbed character, which has been shown to promote nesting and hatching success of the waterfowl using these communities (Deubbert 1969; Schrank 1972; Deubbert & Lokemoen 1976).

Such practices are likely to promote the expansion of Cirsium populations within these plant communities. Effective control of Cirsium by mowing is dependent on early cutting of stems, especially prior to heading. Control is further improved with repeated cuttings (Hansen 1918; Hodgson 1958). Delayed haying, as currently practiced at Malheur NWR, contradicts recommended control measures and often permits mature seed to develop by the time some cutting is begun (personal observation). Non-use results in decreased production and eventually invasion of less desirable species in tall grass prairies (Ehrenreich 1959; Hulbert 1969) and wet-mesic vegetation (Vogl 1964, 1967; Ward 1968; Stachon & Zimdahl 1980).

Thus, a conflict exists between certain management practices and the desire to halt the apparent invasion and thickening of Cirsium stands in susceptible habitats of Malheur NWR. Wildlife management objectives limit the use of standard Cirsium control measures and, therefore, alternative methods are needed.

## STUDY AREA

## Location and Physical Characteristics: General

Malheur NWR, one of the largest wildlife refuges (72,219 ha) in the contiguous United States, is located about 40 km south of Burns in Harney County, southeastern Oregon (Fig. I.1). The refuge lies within Harney Basin, a broad structural and physiographic depression (Walker & Swanson 1968) which forms the northwestern portion of the Basin and Range Province of the western United States (Baldwin 1964). Internal drainage terminates in Harney and Malheur Lakes in the northern portions of the refuge. Major watercourses flowing into the basin include the Silvies River and Silver Creek from the Blue Mountains to the north and der Donner und Blitzen River (=Blitzen River) from Steens Mountain to the south. The lower Blitzen River valley occupies the southerly portion of the refuge.

Although northwest trending faults form prominent scarps in and near the refuge boundaries, the bulk of the area is composed of flat to gently rolling landscapes derived from pluvial lake sediments and older Pleistocene alluvium (Piper et al. 1939; Walker & Swanson 1968). Relief in these bottomlands is less than 30 m from the Malheur and Harney Lakes (approximately 1250 m elevation) south to where the Blitzen River enters the refuge.

As is typical of many high elevation basins of the Intermountain region, Harney Basin is characterized by a semiarid climate and moderate to cold temperatures (Gomm 1979). Precipitation and

Fig. I.1. General location of Malheur NWR (inset lower left) and location of study sites within the refuge. Plant communities: Scirpus acutus, Sparganium eurycarpum, Chrysothamnus nauseosus/Elymus cinereus and Elymus triticoides (1); Carex atherodes (2); Juncus balticus and Eleocharis palustris (3); Sarcobatus vermiculatus/Distichlis spicata (4). Location of weather stations: (●).



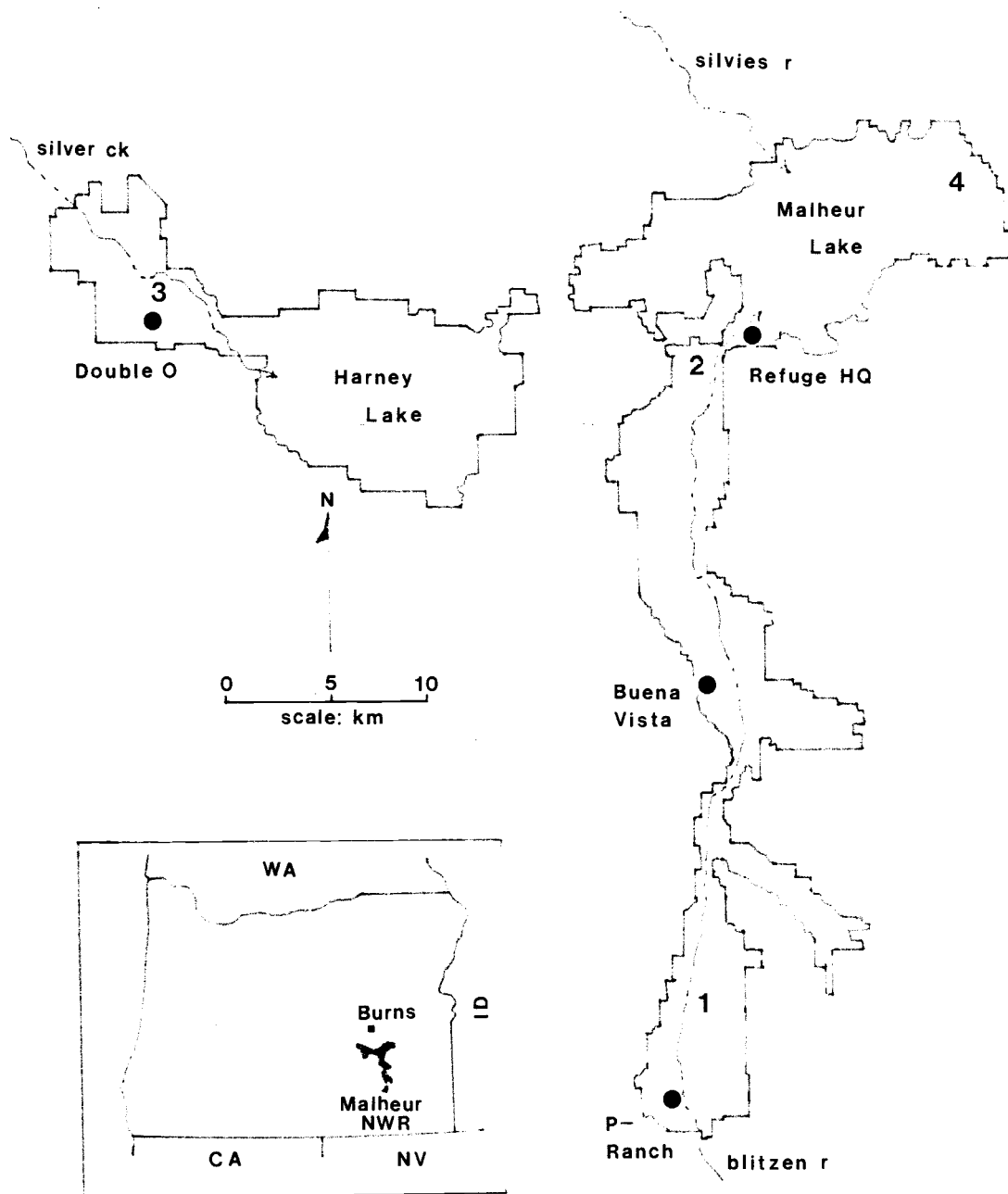


Figure I.1

temperature statistics from several weather stations on Malheur NWR are presented in Table I.3 (see Fig. I.1 for stations locations). Annual precipitation for the basin averages 25-30 cm, but extremes of 10 cm and 50 cm have been recorded (Gomm 1979). The bulk of precipitation occurs in winter and spring, whereas summers are typically dry (Table I.3). Growing seasons are limited by cool spring temperatures, dry summers, and freezing temperatures which generally occur in late August to early September (Gomm 1979).

#### The Wetland-Upland Mosaic: Composition and Management

Vegetation of Malheur NWR is influenced by both natural physiographic and man-made characteristics of the landscape. Historically, extensive marshes have occurred along all major riverine systems of Harney Basin (Griffiths 1902). Beginning in the early 1900's, however, and extending to present times, drastic modifications have been imposed on these areas. Water control has been achieved through river channelization and construction of dams, dikes, and canals. Initially these structures were established to stabilize and improve native hay production (Griffiths 1903). More recently this system has been maintained within the refuge for management of wildlife habitat. Artificial flooding systems over most of the refuge wetlands, exclusive of Malheur and Harney lakes, approximate natural hydrologic regimes in timing and duration, with irrigation beginning after mid-March and drawdown in late July (Hubbard 1975). Depth of

Table 1.3. Long-term precipitation and temperature statistics from weather stations at Malheur NWR, southeast Oregon (National Oceanic and Atmospheric Administration 1983). (See Fig. 1.1 for station locations); ---: no data available; min: minimum; max: maximum.

| Station      | Precipitation (cm) |               |                | Mean Temperature (C) |      |            |      |             |
|--------------|--------------------|---------------|----------------|----------------------|------|------------|------|-------------|
|              | Annual             | Nov-<br>March | April-<br>June | Annual               | Jan  | Jan<br>min | July | July<br>max |
| Headquarters | 23.0               | 10.9          | 7.1            | 6.8                  | -2.9 | ---        | 19.7 | ---         |
| Double O     | 27.6               | ---           | ---            | 8.7                  | -1.1 | -6.5       | 20.2 | 31.3        |
| Buena Vista  | 23.2               | 10.1          | 8.3            | ---                  | ---  | ---        | ---  | ---         |
| P-Ranch      | 30.3               | 14.7          | 10.1           | ---                  | ---  | ---        | ---  | ---         |

inundation, however, is adjusted to facilitate management objectives for individual ponds, marshes, and meadows.

Despite the nearly ubiquitous alteration of riverine and palustrine systems of Malheur NWR, some semblance of a natural state still exists. The long, narrow wetland zones are a continuum of ponds, sloughs, marshes, and wet meadows. Adjacent terrestrial communities finger into the wetlands and frequently form islands of upland vegetation. Thus wetland-upland vegetation of the bottomlands constitutes a complex ecosystem with a mosaic pattern of plant communities and open water.

With the exception of the Pelican Island study site all other study areas occur within the wetland-upland complex (Fig. I.1). The wetland sites are subject to the artificial flood regimes described above; whereas vegetation of the upland sites are subject to the annual vagaries of growing season precipitation, temperatures, soil moisture, etc.

#### Pelican Island

Pelican Island lies off the east shore of Malheur Lake (Fig. I.1). Its main feature is a single north-south oriented dune strand approximately 1.6 km long and varying from 20-75 m in width. On either side of the upraised dune strand are parallel belts of nearly flat terrain, the widths of which are determined by the surface elevation of Malheur Lake. When study sites were located in the summer of 1980, the area west of the strand averaged about 30-40 m wide, while the east side varied between 50-150 m. These belts are

subject to periodic inundation during times of high lake levels. In most years, however, they are dry, although vegetation is strongly influenced by subirrigation from a high water table (personal observation and communication with Malheur NWR staff members).

### Vegetation

Vegetation descriptions for various portions of Malheur NWR are available from three sources. A range survey of the refuge was conducted in the 1950's and early 60's (C. Rouse; unpublished data and maps on file at Malheur NWR, US Fish & Wildlife Service, 1961) which included field identification and mapping of plant communities. Composition and abundance of the common and obvious species were obtained by visual inspection of the fields. Only uplands and wet meadows were described. Other wetlands were mapped simply as "marsh". Although a commendable effort, given the extent and complexity of the task, this work is currently deficient in several respects. Perhaps most important was the failure to classify the communities identified into a structured scheme of vegetation types, and subsequently to provide type descriptions of these units. Further problems arise due to changes in species composition and mapping unit boundaries over the 20+ year period since the survey was completed (personal observation).

Deubbert (1969) briefly describes several upland and emergent plant communities within three 259 ha bird census units on Malheur Lake. Although limited in scope, this is the only such work for emergent communities in this portion of the refuge. Additional

vegetation descriptions are available for a limited area surrounding Harney Lake (Copeland 1979). This work includes a brief discussion of site-vegetation relationships and community type designation for some of the more common species combinations.

Wetland and mesic meadow vegetation varies with depth and length of inundation periods, water and substrate chemistry, and past history of management (Deubbert 1969; Copeland 1979). The most common and abundant aquatic emergent species of deepwater habitats include Scirpus acutus Muhl. (hardstem bulrush) and Typha latifolia L. (common cattail). A variety of other emergents commonly dominate shallower marshes, including: Sparaganium eurycarpum Engelm. (bur-reed), Carex atherodes Spreng. (awned sedge), Carex nebraskensis Dewey (Nebraska sedge), Scirpus americanus Pers. (American threesquare), Scirpus maritimus L. (alkali bulrush), Juncus balticus var. montanus Engelm. (baltic rush), and Eleocharis palustris (L.) Roemer & Schultes (spikerush). In wet meadows and small basins flooded for only short periods in the spring, and in the upland mesic meadows, emergent species give way to the true grasses. The most important of these at Malheur NWR are Distichlis spicata var. stricta (L.) Greene (inland saltgrass), Elymus triticoides Buckley (creeping wildrye), and Poa nevadensis Vasey ex Scribn. (Nevada bluegrass). It is worth noting that, with the exception of Poa nevadensis, all of the above-mentioned species are rhizomatous.

Monotypic stands of aquatic emergents occur in areas ranging from several hectares to patches of a few meters square. Mixed

communities of two to several species, however, are equally common. Typically, the most floristically diverse communities occur on sites flooded to shallow or intermediate depths (<50 cm) and in mesic meadows (personal observation).

Terrestrial communities within the wetland-upland complex appear to be variants of the Artemisia tridentata and Atriplex confertifolia zones described for the region by Holmgren (1972). Based upon the survey by Rouse, the discussions of Deubbert (1969) and Copeland (1979), and personal observations, the following habitat types (sensu Daubenmire 1968) are proposed. These include the most common constituents of the bottomland terrestrial sites occurring at Malheur NWR.

- (1) Sarcobatus vermiculatus/Distichlis spicata var. stricta habitat type (syn.: SAVE/DISP h.t.): Abundance of both Sarcobatus and Distichlis vary widely. Chrysothamnus nauseosus var. consimilis (Greene) Hall & Clem. (salt rabbit-brush) is a common constituent and frequently codominates with Sarcobatus. This habitat type occurs on alkaline soils. Where soils remain moist for long periods in the spring or are subirrigated throughout the summer, Poa nevadensis, Scirpus nevadensis Wats. (Nevada bulrush), and/or Nitrophila occidentalis (Mog.) Wats. (borax weed) occur in the herbaceous union.

(2) Artemisia tridentata ssp. tridentata Nutt./Elymus cinereus

Scribn. & Merr. (Basin wildrye) habitat type

(syn.: ARTR/ELCT h.t.): Although Sarcobatus is usually

present and often abundant, Artemisia and Elymus

become increasingly more important on these sites,

presumably as depth to the water table increases and

soil alkalinity decreases. Chrysothamnus consilimilis

is replaced by Chrysothamnus nauseosus var.

albicaulis (Nutt.) Rydb. (grey rabbitbrush) and

Chrysothamnus viscidiflorus var. viscidiflorus

(Hook.) Nutt. (green rabbitbrush). Fire, at least

temporarily, removes Artemisia from these communities.

The shrub layer is then dominated by Sarcobatus and

species of Chrysothamnus. This habitat type occurs

over large areas of the terrestrial bottom-lands, and upon

detailed investigation, might be divided into a greater

number of vegetation units. This possibility is supported

by the wide variation observed in herbaceous species

composition. In addition to Elymus cinereus,

Elymus triticoides, Poa nevadensis, Distichlis spicata,

and the annual Bromus tectorum L. (cheatgrass) are common

grasses occurring in these communities.

(3) Elymus triticoides habitat type (syn.: ELTR h.t.): This

is the dominant species/habitat type of most mesic meadows.



These are diverse herbaceous communities in which a variety of other graminoids occur, occasionally as codominants with Elymus, or forming localized, nearly pure patches within the stand. These species include Carex douglasii F. Boott (Douglas sedge), Juncus balticus var. montanus, Poa nevadensis, and Carex aquatilis Wahl. (water sedge).

Forbs appear to achieve greatest diversity and abundance in these communities. The noxious weed Cirsium arvense is an aggressive invader of this habitat type as well as many wet meadow communities. The mesic meadows occupy a transitional position along the moisture gradient from the true wetlands, including wet meadows (syn.: flood meadows), to the xeric shrub-grass communities of the uplands.

(4) Sarcobatus vermiculatus habitat type (syn.: SAVE h.t.):

This is an extreme variant of the SAVE/DISP h.t. and might also be classified as intermediate on the moisture gradient.

The shrub Sarcobatus is the sole dominant of these sites, although Distichlis spicata, Nitrophila, and Suaeda intermedia Watts. (tall suaeda) may also be present in small quantities. These communities occur in small basins and flat areas subject to winter-spring flooding.

Sarcobatus appears to be eliminated on sites permanently inundated through the growing season.

## Study Sites

The rationale behind selection of study sites was to investigate the major community types, and therefore, the common and abundant emergent macrophytes of wetland habitats at Malheur NWR. Additional selection criteria included a minimum stand size of 2 ha in a location that would facilitate safe and efficient controlled burning. Five wetland communities were chosen for study. The logistics of conducting the prescribed fires was simplified in that several of the communities occur together within a field and could, therefore, be burned as one unit.

### Wetlands

All wetland plant communities chosen for study occur as monotypic stands. This has the advantage of both simplifying certain sampling procedures and eliminating the potential of confounding treatment effects and interspecific interactions. Additionally, single-species composition is a common structural feature of wetland communities of Malheur NWR.

The species/communities selected represent the common and abundant aquatic emergents of the refuge. All may be found occurring in both monotypic stands and in mixtures of two to several species. Collectively they occupy the entire range of water depths in marshes and wet meadows, from the deepest habitats supporting emergent vegetation to wet soils of small depressions. Individually, several of these species are found growing over large portions of this

spectrum. All of these emergents are widespread in inland wetlands of the temperate and cold-temperate regions of North America, occurring from low to fairly high elevations in the mountains; and all are commonly found in appropriate habitats of the western United States (Franklin & Dyrness 1972; Barbour & Major 1977; Cronquist et al. 1977).

Each site has been classified according to the wetlands and deep-water system of the United States (Cowardin et al. 1979). All five communities fall within the same category: SYSTEM Palustrine, CLASS Emergent Wetland, SUBCLASS Persistent, WATER REGIME Seasonally Flooded, SOIL Mineral, SPECIAL MODIFIER Impounded. The following wetland communities were studied, and are listed in approximate order of decreasing depth of inundation:

- (1) Scirpus acutus community: location -- Knox Field (T.31S., R.32 1/2 E., S.19); this field was managed for non-use for five years prior to treatment.
- (2) Sparganium eurycarpum community: location -- Knox Field (T.31S., R.32 1/2E., S.19); this field was managed for non-use for five years prior to treatment.
- (3) Carex atherodes community: location -- South Meadow East Field (T.27S., R.31E., S.3, SE 1/4); this field was managed for non-use for three years prior to treatment.

- (4) Juncus balticus community: location -- Martha Lake Field (T.27S., R.31E., S.3, SE 1/4); this field was managed for non-use for three years prior to treatment.
- (5) Eleocharis palustris community: location -- Martha Lake Field (T.26S., R.28E., S.24, SE 1/4, and S.25, NE 1/4); this field was managed for non-use for three years prior to treatment.

#### Uplands

Terrestrial communities selected for study represent common upland habitat types of the wetland-upland complex at Malheur NWR. These communities do not necessarily represent pristine or even high successional-stage examples of their respective habitat types. All three sites have undoubtedly been strongly influenced by extended periods of heavy grazing by domestic livestock at various times (Griffiths 1903), and there is evidence of recent fire in at least one of the sites. Based upon the results of the refuge range survey and personal observations the communities are considered to be appropriate examples of seral community types occurring over large areas of Malheur NWR.

- (1) Elymus triticoides community: this is a mesic meadow site in the ELTR h.t.; location -- Knox Field (T.31S., R.32 1/2E., S.18, E.1/2); this field was managed for non-use for five years prior to treatment.

(2) Chrysothamnus nauseosus var. albicaulis/Elymus

cinereus community: this community is representative of the ARTR/ELCI h.t.; location -- Knox Field (T.31S., R.32 1/2E., S19); this field was managed for non-use for five years prior to treatment.

(3) Sarcobatus vermiculatus/Distichlis spicata var.

stricta community: this is a community within the SAVE/DISP h.t.; location -- Pelican Island (T.25S, R.33E., sections unsurveyed); this area was managed for non-use for three years prior to treatment.

SECTION II

PRESCRIBED BURNING IN WETLAND HABITATS OF  
MALHEUR NATIONAL WILDLIFE REFUGE  
SOUTHEASTERN OREGON

Richard P. Young

Richard F. Miller

Robert E. Martin

PREScribed BURNING IN WETLAND HABITATS OF  
MALHEUR NATIONAL WILDLIFE REFUGE  
SOUTHEASTERN OREGON

SUMMARY

(1) Controlled burns were conducted in wetland habitats at Malheur National Wildlife Refuge during the period 9 December 1980 to 20 October 1981. These were used to evaluate the relationship of various fuel and weather variables on fire behavior; and, to develop fire prescriptions and recommendations for burning in these fuel types.

(2) Vegetation consisted of monotypic communities of emergent macrophytes. Habitats supported fuel matrices with heavy loading of continuous, fine fuels. Fuel height varied due to interspecific differences in growth habit and, as a result of lodging. All burns were conducted during periods of vegetation dormancy, fall to early spring.

(3) Fuel and weather parameters measured included: fuel load, standing fuels and litter height, fuel moisture, air temperature, relative humidity and wind speed. Fire behavior was characterized by measurements of rate-of-spread (ROS) and flame length.

(4) Fuel and weather effects on fire behavior were analyzed by means of stepwise multiple regression. ROS was most strongly influenced by wind speed, and secondly, by fuel height. Models incorporating windspeed alone or in combination with a second

independent variable accounted for >75% of the observed variation in ROS. Flame length was most closely related to fuel height. In contrast to terrestrial communities, relative humidity had little influence on fire behavior.

(5) Burning prescriptions and recommended techniques, suitable for use in these and similar wetland habitats were developed. Combined with the fire behavior models, these should provide useful guides for initiating or extending burning programs in inland wetlands.



## INTRODUCTION

Fire is commonly employed as a management technique to manipulate structure, species composition, productivity and succession of wetland vegetation (Lynch 1941; Vogl 1964, 1973; Wade et al. 1980). The principle objective of controlled burning in marshes is to improve wildlife habitat (Schlichtemeier 1967; Vogl 1967; Ward 1968). Recognizing the potential uses as well as the paucity of information concerning fire in western inland wetlands, research was initiated in 1978 to investigate prescribed burning and fire effects on wetland communities at Malheur National Wildlife Refuge (NWR), southeast Oregon. This paper summarizes results and conclusions pertaining to the management aspects of fire, based upon experimental burns conducted under this program. Specific vegetation responses are considered in separate papers (Sections III and IV).

Objectives of this study were: (1) to evaluate the effects of fuel and weather parameters on fire behavior; and (2) to develop burning prescriptions and techniques suitable for major wetland habitats.

## STUDY AREA

Malheur NWR is located approximately 40 km south of Burns in southeast Oregon. The refuge lies within Harney Basin, a closed watershed occupying the northwestern-most portion of the Basin and Range Province (Baldwin 1964). Dominant features include Malheur and

Harney Lakes in the northern portions of the refuge and the lower Blitzen River Valley, a major watershed of the basin, to the south. With the exception of open waters, the bulk of the refuge consists of flat to gently rolling landscapes (elevation: 1250-1280 m) derived from lake sediments and alluvium (Walker & Swanson 1968). Typical of high elevation basins of this region, the climate is semiarid with warm, dry summers and moist, moderate-cold winters (Gomm 1979). Annual precipitation averages 25-30 cm over the refuge.

Riverine and palustrine systems occupy the bottomlands draining into Malheur and Harney Lakes. These are typically long, narrow wetland zones composed of a continuum of ponds, sloughs, marshes and wet meadows. Adjacent terrestrial communities finger into the wetlands, frequently forming islands of upland vegetation. Wetland-upland vegetation of these areas constitute a complex ecosystem with a mosaic pattern of plant communities and open water.

Wetland habitats in these areas are subject to artificial flooding regimes. Water control has been achieved through river channelization and construction of dams, dikes, and canals. Initially established to improve native hay production (Griffiths 1903), this system is currently maintained for the management of wildlife habitat. Artificial flooding systems over most of the refuge wetlands, exclusive of Malheur and Harney lakes, approximate natural hydrologic regimes in timing and duration, with irrigation beginning after mid-March and drawdown in late July (Hubbard 1975). Depth of inundation,

however, is adjusted to facilitate management objectives for individual ponds, marshes, and meadows.

### Study Sites

Five wetland habitats were chosen for evaluating fire behavior. Sites are representative of the common and abundant wetland communities and, therefore, the fuel types occurring in the wetland-upland complex at Malheur NWR. Criteria in locating sites included a minimum stand size of 2 ha, in an area that would facilitate safe and efficient controlled burning.

These sites support monotypic communities of emergent macrophytes. An advantage of this feature, with respect to studies of fire behavior, is the simplification of fuel matrix characteristics. Collectively, the species found at these sites occupy the range of water depths in marshes and wet meadows, from the deepest habitats capable of supporting emergent vegetation to wet soils of small depressions. It was anticipated these characteristics, in addition to the widespread occurrence of the emergent species selected, would extend the potential usefulness and application of our findings to other regions. These habitats are probably best and most simply defined by the emergent species they support; these include: Scirpus acutus, Sparganium eurycarpum, Carex atherodes, Juncus balticus and Eleocharis palustris (see Section III for a detailed discussion of plant community structure and dynamics).

## METHODS

### General Approach

During the period 9 December 1980 to 20 October 1981, eight prescribed burns were conducted to evaluate fire behavior in wetland habitats at Malheur NWR. A number of variables were measured prior to, during, and after each burn, with the objective of quantifying both fire behavior and site parameters that influence fire behavior. This information was subsequently used to evaluate the relationship between independent variables, that is, the site parameters we monitored, and response variables that describe the observed fire behavior. Ultimately we wanted to produce one or more simple models, based upon the relationship of fire behavior to the site parameters, that could be employed by land managers interested in using fire as a wetland management tool.

We measured independent site variables pertaining to properties of the fuel type and weather at the time of the burn. Fuel variables included: (1) fuel load--the quantity of fuels per unit area that potentially may be combusted in a fire; (2) fuel height--a descriptor of the arrangement or structure of the fuels matrix; (3) litter height--similar to fuel height, but pertaining to that portion of the fuels near the ground surface, and which are therefore typically denser in terms of biomass per unit volume; and, (4) fuel moisture--reported as the percent of water per unit oven-dry weight of fuels.

Additionally, we measured the following weather variables: (1) air temperature; (2) relative humidity; and (3) windspeed.

### Sampling

#### Fuels

The fuel matrix was described by measurements taken on the variables fuel load, fuel height, litter height, and fuel moisture. The following procedures were employed in obtaining these data.

(1) Fuel loading was measured by sampling total aboveground biomass. Vegetation on these habitats is composed solely of herbaceous species and, therefore, harvested samples of these materials estimate the total fuel load present at a site. Prior to a burn (usually 2-10 days preceeding) a series of quadrats were located throughout the fuel type, harvested to ground level, and collected for oven-drying (48 h @ 60 C) and weighing ( $\pm 0.1$  g). Quadrats, 0.5 x 0.5 m, were systematically located (2-5 m intervals) along randomly located transects. Sample size varied within each habitat, but in all cases were calculated to provide estimates of fuel loads within 10% of the population mean, in 90% of all cases. Sample sizes were determined using Stein's two-stage procedure (Steel & Torrie 1980) with a variance term provided by previous sampling of herbage production (see Appendix A).

(2) Fuel height and litter height were measured at 1-2 m intervals along the above transects. High variability in these parameters precluded sampling at intensities sufficient to provide the

level of accuracy/precision applied to fuel load measurements. However, post-sampling calculations indicated all estimates were +20%, or better, in 90% of all cases.

(3) Fuel moisture content was determined from samples collected immediately prior to ignition of an experimental unit. Samples were stored in airtight cans for subsequent drying and weighing (48 h @ 60 C). Data are expressed as the percent water per unit oven-dry weight of fuels. Twenty samples were collected at each fuel type from paced transects across the burn area. This procedure provided estimates of fuel moisture equal to or better than +20% of the true mean in 80% of all cases.

#### Weather

Weather measurements were collected prior to, periodically during, and immediately after each burn. Air temperature, relative humidity (sling psychrometer), and wind speed (hand held gauge supplied with belt-weather kits, held at approximately 2 m height) were measured at 15-20 minute intervals (approx.). This provided 5-10 sets of measurements over the course of each fire.

#### Fire behavior

Characteristics of fire behavior were measured by two means: rate-of-speed (ROS) and flame length. The relationship of these variables to fire intensity, and their potential usefulness in evaluating fire effects relative to fire behavior have been emphasized by Rothermel & Deeming (1980).

ROS was measured by timing the movement of a flaming front through fuels and subsequently measuring the distance after passage of the flames. Both heading and backing fires were considered. These observations, although quantitatively less precise than other methods (Britton et al. 1978), permitted us to obtain a larger number of observations within a fuel type in which fire behavior was, as yet, unpredictable.

Flame length was estimated by visual assessment from a vantage point as close to the flaming front as was deemed safe. Again, we were willing to accept less precision in our estimates for a larger number of observations.

### Controlled Burns

#### Firelines

We utilized a variety of natural and constructed firelines, in part to permit an evaluation of their effectiveness. Naturally occurring fuel breaks included the edges of ponds, sloughs and canals, both with and without water present at the time of burning.

We constructed firebreaks to limit the size of burn units where natural breaks were unavailable, or to connect natural breaks (eg. between two or more canals, ponds, etc.). Two methods were used, depending on the size and density of fuels.

In lighter fuels such as the Carex, Juncus and Eleocharis types, a rotary mower (=rotobeater) was used to cut and scatter standing fuels and litter. Fuel density often dictated a need for several

passes, cutting to successively lower stubble height. This procedure typically resulted in a final stubble height of about 10-15 cm.

Heavier fuels such as Scirpus required the use of a crawler tractor to open firebreaks. This method of fireline construction provided the additional safety feature of exposing mineral soil in those areas where the potential of peat burns existed.

The constructed firebreaks were 2-4 m in width. Natural breaks varied from 2 m to over 20+ m.

#### Ignition methods

All burning was initiated by igniting test burns to permit us the opportunity of evaluating fire behavior before burning in the main unit. Frequently, isolated patches of fuels were available. In several cases test burn plots needed to be constructed. We placed these on the downwind edge of burn units (based upon expected direction of the prevailing seasonal winds), using the above methods to isolate the test burn from the main burn unit.

Ignition was conducted in all of these burns by means of hand-held drip torches. All burning was initiated on the downwind boundary of units. Once backing fires had opened a sufficiently wide blackline, strip headfires were used, progressively widening the strips. When judged safe to proceed, the remainder of a unit was headfired.

#### Analysis

The relationship of fire behavior to fuel and weather variables was analyzed by stepwise multiple regression. This procedure was



used to evaluate the relative importance of each independent site variable both alone and in combination with the other parameters. Our approach was to identify those parameters which most strongly influence fire behavior in wetland fuels; and secondly, to provide a quantitative relationship between the "best" independent variables and expected fire behavior. By this process we hoped to produce simple models, incorporating as few site parameters as possible, which could be used by wetland habitat managers to aid in planning and conducting controlled burns.

Backward stepwise regression techniques were used in these analyses. The decision to include additional variables was based upon the criteria: (1) minimize the number of independent variables, preferably  $\leq 2$ , while accounting for a significant amount of the variation in the response variables; (2) maximizing  $R^2$ , the multiple correlation coefficient; and (3) minimizing MS, the residual mean sums of squares (Neder & Wasserman 1974). The first of these criteria was emphasized in the final selection of models.

## RESULTS AND DISCUSSION

Fuels in these wetland habitats are characterized by heavy fuel loads of continuous, fine fuels (Table II.1). High productivity and tall stature may result in lodging or layering of fuels and, therefore, highly variable height structure. Fire behavior was observed to vary significantly in response to layering of fuels typically through rapid reduction of both ROS and flame length

Table II.1. Preburn fuels and weather conditions during experimental prescribed fires conducted at Malheur NWR, 8 December 1980 to 20 October 1981.

| Community                    | Burn date        | Preburn fuel load<br>(g m <sup>-2</sup> )<br>$\bar{x}$ | range     | Fuel<br>height<br>(cm) | Litter<br>height<br>(cm) | Fuel<br>moisture<br>(%) | Temperature<br>(C) | Relative<br>humidity<br>(%) | Wind<br>speed<br>(km h <sup>-2</sup> ) |
|------------------------------|------------------|--|-----------|------------------------|--------------------------|-------------------------|--------------------|-----------------------------|--|
| <u>Scirpus acutus</u>        | 9 December 1980  | 1817   | 1062-2402 | 184                    | 74                       | 4.6                     | 3-5                | 27-34                       | 2-16                                   |
|                              | 7 April 1981     | 1707   | 887-2208  | 122                    | 66                       | 5.8                     | 4-9                | 31-35                       | 6-10                                   |
| <u>Sparganium eurycarpum</u> | 9 December 1980  | 860  | 514-1280  | 39                     | 19                       | 16.3                    | 3-5                | 27-34                       | 2-16                                   |
|                              | 7 April 1981     | 825  | 450-1312  | 26                     | 17                       | 5.8                     | 4-9                | 31-35                       | 6-10                                   |
| <u>Carex atherodes</u>       | 18 December 1980 | 1120   | 617-1519  | 20                     | 15                       | 17.5                    | 4-8                | 62-77                       | 2-6                                    |
|                              | 6 April 1981     | 1136   | 760-1468  | 18                     | 15                       | 5.9                     | 5-9                | 27-35                       | 10-19                                  |
| <u>Juncus balticus</u>       | 20 October 1981  | 851  | 591-1104  | 29                     | 12                       | 3.5                     | 16-23              | 13-17                       | 3-16                                   |
| <u>Eleocharis palustris</u>  | 20 October 1981  | 550  | 389-805   | 12                     | 6                        | 3.7                     | 16-23              | 13-17                       | 3-16                                   |

(personal observation). Therefore, ROS and flame length measurements were stratified on this feature and related to preburn fuel height for lodged or upright fuels. A summary of preburn fuel characteristics is given in Table II.1.

Burning was conducted with a variety of weather conditions (Table II.1). Winter and early-spring burns took place with low temperatures. Additionally, variable snow cover (<5 cm depth) was present around stem bases of upright fuels, and as small patches atop some lodged fuels (<5% total cover on any burn unit). These conditions are reflected in the low to moderate ROS, flame lengths and fire intensities of these burns (Table II.2). Sparganium and Carex winter burns produced the lowest fire intensities of the burns we conducted; and, based upon observed fire behavior and fuels consumption, it was felt these conditions were near the low limit of intensity necessary to achieve acceptable results. This was especially true in the Carex burn where the percentage of lodged fuels was greatest (45% of the burn unit). Despite these observations, all experimental burns were judged successful, and postburn fuel samples supported this subjective evaluation (Table II.2).

Fall burns in Juncus and Carex were conducted with the highest temperatures and lowest relative humidities (Table II.1). Again, this is reflected in the fire behavior measurements (Table II.2). Although there was no difficulty with fire control, it was felt these

Table II.2. Fire behavior and effects on fuels during experimental prescribed fires conducted at Malheur NWR, 9 December 1980 to 20 October 1981.

| Community                    | Burn date        | Rate of Spread<br>(m min <sup>-1</sup> ) |          | Flame length<br>(m) |          | Fire intensity<br>(kW m <sup>-1</sup> ) |          | Postburn residual<br>fuels<br>(g m <sup>-2</sup> ) |        | Reduction<br>(%) |
|------------------------------|------------------|--|----------|---------------------|----------|---|----------|--|--------|------------------|
|                              |                  | Headfire                                 | Backfire | Headfire            | Backfire | Headfire                                | Backfire | $\bar{x}$  | range  |                  |
| <u>Scirpus acutus</u>        | 9 December 1980  | 5-30                                     | 1-1.5    | 3-10                | 1-2      | 2516-5774                               | 403-925  | 309  | 27-568 | 83               |
|                              | 7 April 1981     | 20-25                                    | 1-2      | 3-4                 | 1-2      | 6354-15947                              | 406-1026 | 154  | 12-328 | 91               |
| <u>Sparganium eurycarpum</u> | 12 December 1980 | 2-10                                     | 1-1.5    | 1-5                 | 1        | 707-1805                                | 192-490  | 206  | 21-485 | 76               |
|                              | 7 April 1981     | 20-40                                    | 1-2      | 1-5                 | .5       | 4546-13412                              | 207-610  | 124  | 8-236  | 86               |
| <u>Carex atherodes</u>       | 18 December 1980 | 2-5                                      | .75-1.5  | 1-2                 | 1.5      | 620-1549                                | 211-528  | 246  | 14-453 | 78               |
|                              | 6 April 1981     | 10-15                                    | 1.5      | 2-5                 | .5-1     | 2518-4879                               | 294-570  | 136  | 5-186  | 88               |
| <u>Juncus balticus</u>       | 20 October 1981  | 50-60                                    | 1-2      | 3-6                 | 1.5      | 10131-18924                             | 276-516  | 68   | 0-128  | 92               |
| <u>Eleocharis palustris</u>  | 20 October 1981  | 20-30                                    | 1-1.5    | 1.5-3.5             | 1-1.5    | 3031-6272                               | 152-314  | 33   | 0-96   | 94               |

conditions would have been inappropriate for heavier fuel loads, as in Scirpus habitats.

Postburn fuels sampling provided an estimate of the quantity of fuels combusted in each fire (Table II.2). This information was used to calculate fire intensity for heading and backing fires. The procedures suggested by Alexander (1982) were followed, utilizing a basic value of  $18700 \text{ kW kg}^{-1}$  for low heat of combustion. Characterizing fire behavior in this manner permits a quantitative comparison with results from other fires. To date, however, this is the first application of this approach in wetland fuels. In fact, few published accounts of wetland burns have reported details of fuels, weather, and/or fire behavior to permit even general comparisons.

Schlichtemeier (1967) reported weather conditions during four days of controlled burning in marshes of the Nebraska Sandhills. Burns were conducted with temperatures  $-8$  to  $+6$  C, relative humidities 52-72%, and winds  $11-37 \text{ km h}^{-1}$ . The objective of these burns was to improve waterfowl habitat by removing old, dense stands of Phragmites communis and Scirpus sp. All burns were conducted in winter when marshes were frozen and surrounding rangelands were covered by several centimeters of snow. This resulted in some unburned patches and strips; but in burned areas residual plant materials were consumed to a 5-15 cm stubble height over the snow. Densities of Phragmites and Scirpus were reduced 85% and 60%, respectively. Schlichtemeier concluded that despite considerable variation in relative humidity and temperature over the four days of burning, little variation in

vegetation "combustibility" was observed. Winter burns conducted at Malheur appear quite comparable to those described by Schlichtemeier, in terms of fuel and weather conditions and the resulting fire behavior.

Burning is utilized at Delta Marsh in south-central Manitoba for waterfowl habitat improvement (Ward 1968). Spring burning is employed to create greater nesting edge, to thin dense stands of marsh vegetation, and to remove invading woody species. Due to moist substrates these burns remove accumulated graminaceous vegetation but do not affect regrowth. Firing is conducted on bright, warm days after several preceding days of similar weather, and with wind speeds less than  $32 \text{ km h}^{-1}$ . Edges of bays and creeks and roads have been used as natural firelines. Mechanically cut and cleaned firelines of 6-9 m width are used by backfiring away from the lanes before igniting headfires. Although conducted with apparently warmer temperatures than spring burns at Malheur, the conditions reported by Ward provide useful guidelines to test, and thereby extend the range of weather conditions we evaluated.

A preliminary study of prescribed burning was conducted in wet meadow vegetation at Malheur NWR (Britton et al. 1980). Preburn fuel load was  $7100 \text{ kg ha}^{-1}$  of continuous fine fuels. The burn was conducted in November 1978. Based upon these results Wright & Bailey (1982) suggest prescribed fires in this vegetation type be conducted with air temperatures of 10-18 C, relative humidity 30-40%, and 0-3  $\text{km h}^{-1}$  winds. Our findings extend the limits presented by these

authors; but specific recommendations will be held until further discussion has been presented.

### Fire Models

Regression analyses were conducted using the entire data set of site parameters and behavior measurements, and with a variety of combinations of one to several habitats. ROS was consistently most strongly influenced by wind speed (Table II.3). Field observations, including burns conducted outside the context of this study, support these analyses. We found the only restrictions to successful burning were moisture on the surface of fuels and absence of a steady wind. Wind speed alone accounted for >50% of the variability in ROS for the entire data set. This model could be improved dramatically, however, by removal of either or both the Scirpus and Carex observations (Table II.3). With both these removed, wind alone explained almost 70% of the variation in ROS. The addition of fuel load as a second independent variable to this reduced set increased this value to almost 90%.

ROS in the Carex fuels alone were strongly influenced by fuel height (Table II.3). Again, field observations support these results. We observed extensive lodging of fuels in this habitat, and ROS responded dramatically to this fuel condition by slowing to levels similar to ROS of backing fires.

ROS in Scirpus fuels were also most strongly influenced by wind speed. These fuels are also subject to lodging and, significantly, the

Table II.3. Linear regression models relating fire behavior of strip headfires to fuel and weather parameters measured during experimental prescribed fires conducted at Malheur NWR. All independent variables included in models were significant (.01 level) based upon t-tests. Multiple correlation coefficient ( $R^2$ ) and residual mean square (MS) values are given for each model.

| Dependent variables:  | Independent variables:                  |        |
|---|---|--------|
| rate of spread, ROS ( $\text{m min}^{-1}$ )   | fuel load, LOAD ( $\text{g m}^{-2}$ )   |        |
| flame length, FL (m)  | maximum fuel height, MX HT (M)          |        |
|   | temperature, TEMP (C)                   |        |
|   | fuel moisture, $\text{H}_2\text{O}$ (%) |        |
|   | wind speed, WIND ( $\text{km h}^{-1}$ ) |        |
| (1) All data sets included; n = 66  | $R^2$                                   | MS     |
| ROS = $-6.17 + 3.06$ (WIND)   | .512                                    | 176.92 |
| = $-7.95 + 2.38$ (WIND) + $0.83$ (TEMP)   | .582                                    | 153.78 |
| (2) All data sets except <u>Carex atherodes</u> ; n = 54                              |   |        |
| ROS = $-10.63 + 3.89$ (WIND)  | .729                                    | 103.34 |
| (3) All data sets except <u>Carex atherodes</u><br>and <u>Scirpus acutus</u> ; n = 42 |   |        |
| ROS = $-13.12 + 4.12$ (WIND)  | .690                                    | 134.40 |
| = $-62.06 + 3.95$ (WIND) + $0.07$ (LOAD)  | .897                                    | 45.87  |
| (4) <u>Carex atherodes</u> data set; n = 12   |   |        |
| ROS = $77.89 - 3.73$ (MX HT)  | .844                                    | 2.91   |
| (5) <u>Scirpus acutus</u> data set; n = 12  |   |        |
| ROS = $-3.63 + 2.82$ (WIND)   | .757                                    | 23.12  |
| = $13.56 + 2.76$ (WIND) - $0.11$ (MX HT)  | .890                                    | 11.26  |
| (6) All data sets included; n = 71  |   |        |
| FL = $2.69 + 0.02$ (MX HT)  | .387                                    | 2.99   |
| = $-0.01 + 0.24$ (MX HT) + $.03$ (LOAD)   | .540                                    | 2.34   |
| (7) All data sets except <u>Carex atherodes</u><br>and <u>Scirpus acutus</u> ; n = 47 |   |        |
| FL = $1.13 + 0.10$ (MX HT)  | .304                                    | 1.41   |
| = $-2.04 + 0.13$ (MX HT) + $0.24$ (WIND)  | .778                                    | 0.46   |



second variable to enter this model was fuel height. Despite this, a satisfactory model could not be produced from the combination of the Scirpus and Carex data sets.

Flame length was related to fuel height, but by itself, this variable could account for <40% of the observed variance (Table II.3). Wind speed improved predictability of flame length to 53%; and by removing the Scirpus and Carex observation, this two-variable model accounted for >75% of the variation in flame length.

#### Recommendations and General Comments

It is noteworthy that RH was not a significant variable in any of the regression models we evaluated. This is in contrast to most terrestrial fuel types where fuel moisture is closely related to relative humidity (Countryman 1964, Mobley et al. 1973). In grassland fuels, relative humidity is commonly employed to estimate thresholds for moisture of extinction (Mobley et al. 1973) and, as an indicator of spotting potential due to airborne firebrands (Wright & Bailey 1982). The lack of importance of relative humidity with respect to fire behavior in wetland fuels is likely related to fuel surface properties. Emergent species are essentially "water-tight" (Mason 1957), a necessity in this environment. Therefore, unless the epidermis is broken, changes in relative humidity appear to have little influence on the uptake or loss of moisture in these fuels.

### Fire prescription

Burning prescriptions should be specific to fuel types. In wetland habitats, however, differences in species composition are not coincident with significant changes in fire behavior, with the exception of fuels that are strongly lodged and in the heaviest fuel types (eg. Scirpus). In general, we found burning objectives could be accomplished with a wide range of weather conditions. The following specific recommendations are based upon the experience of conducting and observing fires during the course of this study, additional burns conducted during this period (but not within the context of this project), and results of regression analyses of fire behavior. There is a high degree of similarity among these recommendations, the principle differences based upon variations in fuel quantity (load) and fuel height, especially with respect to lodging.

Scirpus habitats are broken out due to heavy fuel loads and, secondarily, on fuel height. Both characteristics increase fire intensity to the extent that firelines must be wider than for other fuels. We achieved satisfactory results with winter-spring burns and do not feel burning with warmer, drier conditions warrant the likely increase in difficulty of controlling such fires.

Fuel type: Scirpus acutus.

Season: Late fall to early spring.

Wind:  $<20 \text{ km h}^{-1}$ ; steady winds are needed when  $<5 \text{ km h}^{-1}$ .

Precipitation: Any time after most recent rainfall when the surface of fuels are dry; and, after a snowfall, once snow has melted or settled below the fuel canopy.

Temperature: 0-15 C; with low temperatures, minimum wind speed=4-6 km h<sup>-1</sup>.

Burning pattern: Backfire downwind line; strip headfire to develop a backline of 30-50 m, depending on wind speed; headfire the unit. Center or ring burn with wind <2 km h<sup>-1</sup>.

Fireline: Any natural or constructed fuelbreak, 4-6 m wide. Mechanically cut lines may require backfiring from a wetline (Martin et al. 1977). Presence of organic soils require trenching to mineral soil or the ability to flood the burn unit if peat burns develop.

Fuel type: Upright wetland fuels, other than Scirpus acutus.

Season: Late fall to early spring.

Wind: <25 km h<sup>-1</sup>; steady winds are needed when <5 km h<sup>-1</sup>.

Precipitation: Any time after most recent rainfall when the surface of fuels are dry; and, after a snowfall, once snow has melted or settled below the fuel canopy.

Temperature: 5-25 C.

Burning pattern: Backfire downwind line; strip headfire to develop a backline of 10-25 m, depending on wind speed; headfire the unit. Center or ring burn with wind <2 km h<sup>-1</sup>.

Fireline: Any natural or constructed fuelbreak, 2-3 m wide.

Mechanically cut lines may require backfiring from a wetline.

Presence of organic soils require trenching to mineral soil or the ability to flood the burn unit if peat burns develop.

Fuel type: Lodged wetland fuels, other than Scirpus acutus.

Season: Late fall to early spring.

Wind: 6-30 km h<sup>-1</sup>.

Precipitation: Any time after most recent rainfall when the surface of fuels are dry; and, after a snowfall, once snow has melted or settled below the fuel canopy.

Burning pattern: Backfire downwind line; strip headfire to develop a backline of 10-25 m, depending on wind speed; headfire the unit. Center or ring burn with wind <2 km h<sup>-1</sup>.

Fireline: Any natural or constructed fuelbreak, 2-3 m wide.

Mechanically cut lines may require backfiring from a wetline.

Presence of organic soils require tranching to mineral soil or the ability to flood the burn unit if peat burns develop.

Although directly applicable to wetlands at Malheur NWR, these prescriptions should be applicable to a wide range of inland wetland habitats. When initiating a burning program, extra caution should be used. Start with small burns while developing experience and evaluating fire behavior specific to local habitats. Combinations of

high temperature and winds, and low humidities should be avoided until a thorough understanding of fire behavior has been achieved.

The regression models should be employed solely as guides in planning burns, and not for the purpose of setting specific limits or burning prescriptions. They serve best to point out the relationship and relative importance of habitat variables to expected fire behavior; and, usage should be limited to that role.

SECTION III

FIRE EFFECTS ON WETLAND VEGETATION OF  
MALHEUR NATIONAL WILDLIFE REFUGE  
SOUTHEASTERN OREGON

Richard P. Young

Richard F. Miller

FIRE EFFECTS ON WETLAND VEGETATION OF  
MALHEUR NATIONAL WILDLIFE REFUGE  
SOUTHEASTERN OREGON

SUMMARY

(1) Effects of dormant season burning (fall, winter and early-spring) and non-use on monotypic communities of Scirpus acutus, Sparganium eurycarpum, Carex atherodes, Juncus balticus and Eleocharis palustris were investigated in freshwater wetlands at Malheur National Wildlife Refuge, southeast Oregon. Vegetation response was evaluated in terms of populations of shoots for the parameters: shoot height and growth rate, aboveground standing crop, residual organic materials, shoot density, shoot weight and reproductive effort. Measurements were taken for two years after burning during the dormant period.

(2) Vegetation response was largely insensitive to season of burning.

(3) Initiation of spring growth, rate of growth and maximal height achieved were similar among treatments, within vegetation types.

(4) Burning increased aboveground standing crop (ASC) for two years in all but the Sparganium community. Early-spring burning of Sparganium increased ASC for one year. There was no difference in ASC on winter burned and non-use Sparganium areas. The relationship of fire intensity, residual organic materials and ASC is discussed.

Dynamics of organic residues suggest a return to preburn, that is, non-use conditions in three to five years after fire.

(5) Burning increased shoot densities for two years in all but the Sparganium community. Burning had no influence on Sparganium shoot density or shoot weight. Shoot weight of other emergent species tended to decrease after fire; although weight reduction was significant only in Scirpus vegetation.

(6) Reproductive effort, measured as the percent of shoots which produced inflorescences, was markedly different among species and in response to fire. Scirpus and Carex increased reproductive effort for one year after burning.



## INTRODUCTION

Inland wetlands have often been considered stages in hydrarch succession which terminate in a terrestrial climax (Valk 1982). Palynological studies, however, have shown wetland vegetation may be more stable over geological time than associated terrestrial communities (Watts & Winter 1966; Watts & Bright 1968). Historical evidence from analysis of pollen and charcoal records indicate fire has a significant role in maintaining inland wetlands (Vogl 1969; Davis 1979). Observations of naturally occurring marsh fires (Viosca 1928, 1931; Lynch 1941) support the notion that fire is an important agent influencing community structure and functional processes in wetland ecosystems (Vogl 1977).

The beneficial use of fire in wildland management is dependent on understanding fire effects in specific kinds of ecosystems (Wright & Bailey 1982). Information on fire effects and management applications in North American wetlands concerns vegetation and wildlife in Florida (Robertson 1962; Klukas 1973; Vogl 1973; Bancroft 1976; Wade et al. 1980) and Gulf Coast marshes (Lynch 1941; Singleton 1951; Givens 1962; Hoffpauer 1968; Perkins 1968; Oefinger & Scifres 1977). Whereas some studies have dealt with marshes of the midwestern United States (Schlichtemeier 1967; Vogl 1964, 1967) and Canada (Ward 1968), little is known about controlled burning and fire effects in inland wetlands of western North America (Smith & Kadlec 1985).

In this study we address the general question of the effects of dormant season burning in aquatic emergent vegetation of inland

wetlands. The approach we employ compares surface burns and non-use to test the hypothesis that fire has no effect on subsequent plant growth rates, population structure, aboveground standing crop (ASC) and floral reproductive effort of several wetland species/communities. Surface fires are those which remove only the aerial stems and surface accumulations of organic debris; as opposed to peat or ground fires which consume soil organic matter and plant root systems (Lynch 1941). Non-use denotes management which precludes fire or agronomic practices (eg. hay production, livestock grazing), and, therefore, provides a reference for comparison with other land use practices.

#### STUDY AREA

Studies were conducted at Malheur National Wildlife Refuge (NWR), southeastern Oregon, USA. Malheur is one of the largest wildlife refuges (72, 219 ha) in the contiguous United States, providing critical habitat for nesting and migratory waterfowl and seasonal or yearlong habitat for numerous other wildlife species.

Malheur lies within an internally drained basin (elevation: 1250-1280 m) and consists primarily of flat to gently rolling lowlands derived from pluvial lake sediments and alluvium (Walker & Swanson 1968). Cold desert shrub vegetation of upland sites, dominated by species of the Asteraceae and Chenopodiaceae families, reflect the regionally semiarid climate (mean annual precipitation=25-30 cm) (Franklin & Dyrness 1972; Gomm 1979). Extensive areas of wetland vegetation occur, however, in and around the terminal lakes and

throughout the riverine and palustrine zones of watercourses entering the basin.

The wetland zones at Malheur present a mosaic of ponds, sloughs, marshes and wet meadows. Dominant emergent plant species of these areas include Scirpus acutus Muhl. (hardstem bulrush), S. americanus Pers. (American threesquare), S. maritimus L. (alkali bulrush), Typha latifolia L. (common cattail), Sparganium eurycarpum Engelm. (bur-reed), Carex atherodes Spreng. (awned sedge), C. nebraskensis Dewey (Nebraska sedge), Juncus balticus var. montanus Engelm. (Baltic rush) and Eleocharis palustris (L.) Roemer & Schultes (spikerush) (Deubbert 1969a; Copeland 1979). Each of these species are rhizomatous (Cronquist et al. 1977), and all rely heavily on vegetative growth of these organs to maintain or increase their respective dominance within wetland communities (Mason 1957).

Water control measures were initiated in the early 1900's to improve native hay production in bottomland areas of Malheur (Griffiths 1903). Today, artificial flooding systems are maintained for the purpose of managing wildlife habitat. Natural hydrologic regimes are approximated with respect to timing and duration, with irrigation beginning about mid-March and drawdown in late July (Hubbard 1975).

## MATERIALS AND METHODS

### Field Methods

Five monotypic vegetation types were studied: Scirpus acutus, Sparganium eurycarpum, Carex atherodes, Juncus balticus var. montanus and Eleocharis palustris. Treatments consisted of winter and early-spring burning and non-use in the Scirpus, Sparganium and Carex communities; and, fall burning and non-use in the Juncus and Eleocharis types. Five sites (=replicates) were located for each vegetation type-treatment combination. All study sites occurred within "patches" of monospecific vegetation of size 1 ha. Additionally, only areas managed for non-use during the previous 3 to 5 year period were selected.

#### Controlled burning

The Scirpus, Sparganium and Carex communities were burned during the 1980-81 dormant period. Excessive flooding delayed burning of the Juncus and Eleocharis types until the fall 1981 dormant season. Each replicate was treated (burned) by means of strip headfires (Wright & Bailey 1982). A summary of fuels and fire behavior for each burn is given in Table III.1 (see Section II for a more complete discussion of this material).

Fire intensity, the rate of energy release per unit time per unit length of fire front (Byram 1959) was quantified to allow precise comparisons among these burns and with other fire effects studies (Rothermel & Deeming 1980). Calculated values of fire intensity are

Table III.1. Preburn fuel characteristics, fire intensity (headfires) and fuels consumption for controlled burns in wetland vegetation types of Malheur NWR, southeast Oregon.

| Community                    | Burn date        | Preburn<br>fuel load<br>(g m <sup>-2</sup> ) | Fuel<br>height<br>(cm) | Fire<br>intensity<br>(kW m <sup>-1</sup> ) | Postburn<br>residual fuels<br>(g m <sup>-2</sup> ) | Fuels<br>reduction<br>(%) |
|------------------------------|------------------|--|------------------------|--|--|---------------------------|
| <u>Scirpus acutus</u>        | 9 December 1980  | 1817   | 184                    | 2516-5774                                  | 309  | 83                        |
|                              | 7 April 1981     | 1707   | 122                    | 6354-15947                                 | 154  | 91                        |
| <u>Sparganium eurycarpum</u> | 9 December 1980  | 860  | 39                     | 707-1805                                   | 206  | 76                        |
|                              | 7 April 1981     | 825  | 26                     | 4546-13412                                 | 124  | 86                        |
| <u>Carex atherodes</u>       | 18 December 1980 | 1120   | 20                     | 620-1549                                   | 246  | 78                        |
|                              | 6 April 1981     | 1136   | 18                     | 2518-4879                                  | 136  | 88                        |
| <u>Juncus balticus</u>       | 20 October 1981  | 851  | 29                     | 10131-18924                                | 68   | 92                        |
| <u>Eleocharis palustris</u>  | 20 October 1981  | 550  | 12                     | 3031-6272                                  | 33   | 94                        |

for strip headfires (Martin & Dell 1978) and follow the procedures of Alexander (1982), using a basic value of  $18700 \text{ kJ kg}^{-1}$  for low heat of combustion. More generally, these burns were best characterized as moderate ( $<4000 \text{ kW m}^{-1}$ ) to high intensity fires (McArthur & Cheney 1966). All burns produced uniform, steady-moving flame fronts, which consumed  $>75\%$  of surface fuels in all communities.

### Sampling

To test the assumption that treatment areas within a vegetation type were comparable prior to burning, standing crop (current and previous year's components) and stem density were sampled following the 1981 growing season. Ten quadrats  $0.5 \times 0.5 \text{ m}$  were located in each treatment (two quadrats per replicate). Within each quadrat live stems were tallied and biomass was harvested to ground level, by component.

Post-treatment vegetation measurements were made during the 1981 and 1982 growing seasons at the Scirpus, Sparganium and Carex sites, and in 1982 and 1983 at the Juncus and Eleocharis sites. All sampling was conducted utilizing randomization procedures (Steel & Torrie 1980) to locate specific sample points, quadrats and/or transects within a study site, relative to a central, permanently marked point.

Because of the difficulty in identifying individual "plants" of rhizomatous species such as these, each community was treated as a population of vegetative structural units, or ramets. Measurements taken on random samples of stems were used to infer treatment effects on the population of parent plants, the genets (Harper 1977).

Growth rates were determined by measuring stem heights at 10-14 day intervals throughout the growing season. Repeated measurements of marked shoots were deemed impractical due to excessive physical disturbance after only a few visits to a site. Therefore, a new set of shoots were randomly located and measured on each sample date. Fifty stems were measured at each replicate-date: 10 contiguous stems in each of 5 variable-length transects; N=250 stems per treatment per sample date.

Shoot density, ASC, and the percentage of flowering stems were determined immediately following drawdown of floodwaters; typically, in the last week of July. These measurements were taken in eight quadrats of 0.5 x 0.5 m, per replicate. Quadrats were randomly located relative to a permanently marked point, centrally located within each replicate study area. Within a quadrat, the total number of stems and those with inflorescences were counted. Aboveground biomass was then harvested and separated into the categories: (1) current season's growth, including live green and attached dead leaves; and (2) residual organic materials (litter and old standing dead). Cessation of height growth coincided with drawdown of floodwaters in late July (Table III.2), indicating aerial biomass accumulation was greatly reduced or had ceased prior to the harvest.

Based upon Stein's two-stage procedure for determining sample size (Steel & Torrie 1980), the sampling design/intensity used in this study produced estimates of mean population values within +10% at the 90% confidence level for the plant parameters: stem height and

growth rate, density and ASC. High variability in flowering response precluded sampling intensities sufficient to provide similar levels of accuracy and precision with respect to this parameter.

### Analysis

Analyses of variance were conducted to test the assumption of preburn comparability, with regards to live standing crop, residual debris and stem density, among treatments of each vegetation type. It was anticipated that, where pretreatment differences did exist, subsequent analyses would utilize preburn standing crop and/or density as a covariate.

Analyses of variance for factorial experiments (Cochran & Cox 1966) were used to test treatment (burned, non-use), time (years after treatment) and treatment x time effects for stem growth rate, height, density, ASC, surface organic debris and flowering stem density (absolute and relative values). Quadrats sampled within each replicate were treated as subsamples in these analyses. All F tests were conducted using the experimental error term, from which the sampling error had been partitioned. Prior to these analyses each variable was tested for goodness-of-fit to a normal distribution. Where treatment effects were significant ( $P < .05$ ), least significant difference (lsd) tests were conducted for all possible paired comparisons (Steel & Torrie, 1980).

Evaluation of stem growth, including height and rate of growth, was carried out using four time periods, of length 23 to 44 days.



Time periods were delineated on the basis of natural growth phases observed after plotting the seasonal height growth curves. That is, inflection points were used to delimit segments of the curve which display relatively constant slope (=rate of height growth). Both the rate of stem height growth (daily increment) and absolute height, measured on the dates that separate the growth phases, were tested for differences due to treatment effects.

## RESULTS

Flowering stem density was consistently the only vegetation parameter for which data sets were non-normal. This was true even after standard transformations (square root, log, arcsine) were conducted. Differences in  $\chi^2$  values between transformed and untransformed data were small, however, and therefore untransformed data were used in all analyses of variance.

No differences were observed in 1980 in live or dead aboveground standing crop or shoot density ( $P > .05$ ) among study sites within their respective community types (Appendix A). Therefore, tests of posttreatment vegetation differences did not make use of pretreatment values as covariates to adjust for site differences.

### Shoot Development: Height and Rate of Growth

Trends in shoot development were similar among treatments within each of the five plant communities. Plots of shoot height for each growing season yielded growth curves approximating a sigmoid

TIME x HEIGHT relationship for all five species. Growth curves were, therefore, visually divided into the following developmental phases: (1) pre-emergence -- an initial period of slow growth, prior to emergence above irrigation levels; (2) rapid growth -- stems of all species began to exceed water levels in the first to third weeks of May, at which time rate of height growth increased dramatically; and, (3) vegetative maturity -- late July to mid-August marked the transition to a period of little or no subsequent height growth, indicating vegetative maturity and likely, the cessation of vegetative biomass accumulation. These phases are visible in the plant heights and growth rates given in Table III.2.

Tests of treatment differences for height and growth rate were conducted using values from dates coinciding with the (approximate) transition from one growth phase to another and the daily growth increment of the interim period, respectively. Height or growth rate did not differ ( $P > .05$ ) for dates or growth periods tested for either the first or second years, among treatments of any community type (Table III.2; data presented only for the first year after treatment).

#### ASC and the Dynamics of Residual Organic Materials

Burning, irrespective of season, increased ASC ( $P < .05$ ) for two years in all but the Sparganium community (Fig. III.1). Within communities ASC did not respond in response to season of burning (Fig. III.1, (a)-(c)). Spring burning in Sparganium increased ASC relative to non-use in 1981, the first year after treatment; whereas, ASC of

Fig. III.1. Aboveground standing crop (ASC) and residual organic materials of wetland plant communities, for 2 yr following winter (W) and spring (S) burning, fall (F) burning and non-use (N) treatments. Within a community type, treatment responses with the same letter superscript were not different ( $P > .05$ ). W,S,N/1981-1982 treatments: (a) Scirpus acutus, (b) Sparganium eurycarpum, (c) Carex atherodes communities. F,N/1982-1983 treatments: (d) Juncus balticus and (e) Eleocharis palustris communities.

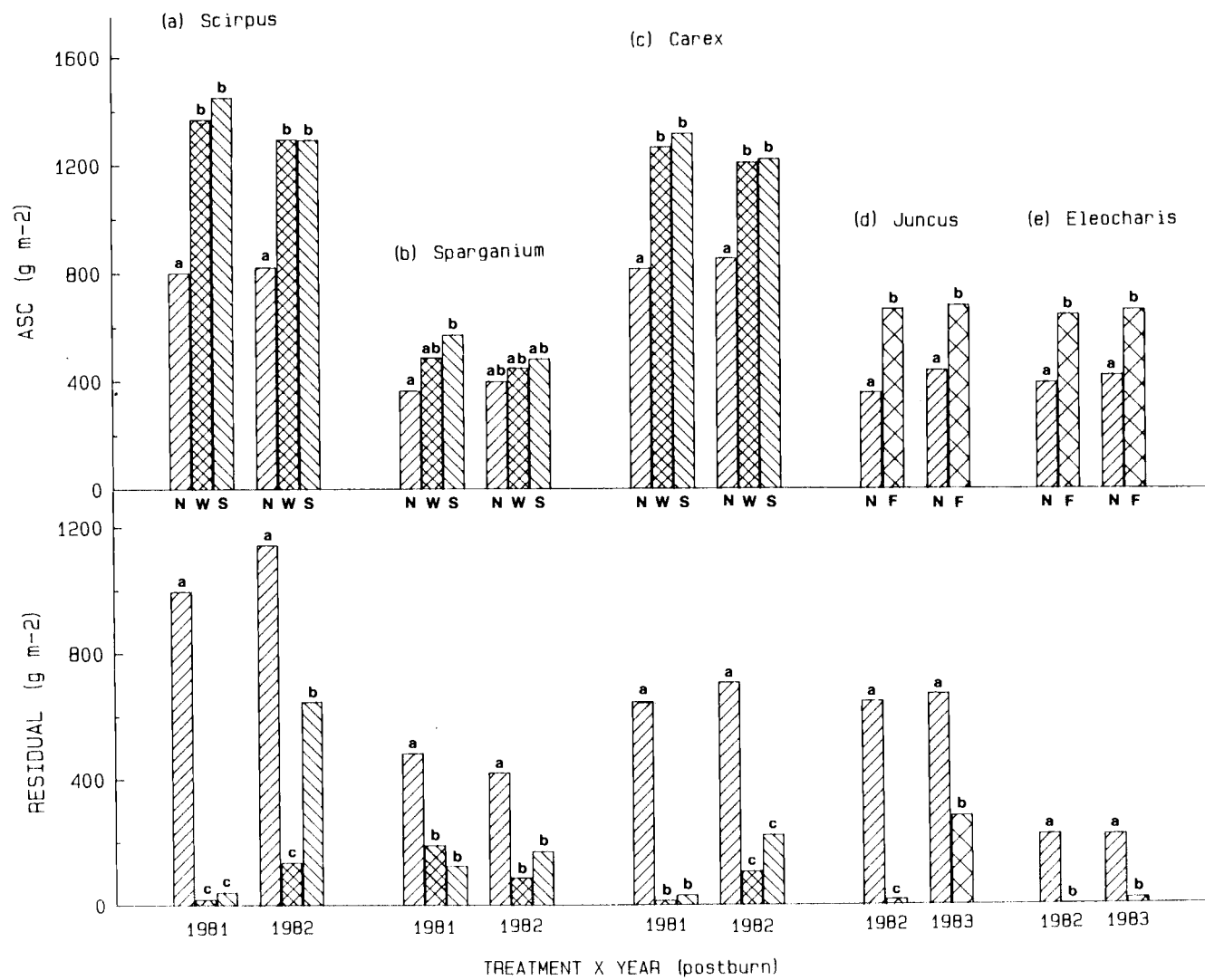


Figure III.1

winter burn sites was not different from non-use at this time (Fig. III.1). However, by the second year ASC was similar ( $P > .05$ ) among Sparganium treatments.

Burning and subsequent decomposition and/or off-site transport dramatically reduced residual organic materials (Fig. III.1). Organic residues increased the second year after burning in Scirpus, Carex and Juncus vegetation; however, these initial accumulations were insufficient to eliminate the differences in biomass of residues between burned and non-use plots. Two years after burning, residues in Eleocharis vegetation (Fig. III.1(e)) were not different from the previous year and were still much lower than non-use. Similar results were observed for burned Sparganium sites, with the exception of winter burning, for which second year residues were intermediate but not different from either spring burning or non-use (Fig. III.1(b)).

#### Shoot Density and Shoot Weight

Burning increased shoot density over non-use for two years ( $P < .05$ ) in all but the Sparganium community type (Fig. III.2, (a)-(e)). Furthermore, stem densities of burned plots did not decrease from year 1 to year 2. Burning had no effect on density of Sparganium shoots (Fig. III.2(b)).

Mean shoot weight did not differ among treatments of any community, with the exception of the Scirpus type, where shoots in non-use areas were significantly heavier (Fig. III.2, (a)-(e)). Carex,

Table III.2. Shoot height and rate of height-growth in wetland communities following seasonal burning and non-use treatments. Only results of the first postburn growing season are shown. Shoot heights correspond with dates delimiting periods of relatively constant growth rates; which are thus considered to represent natural phases of homogeneous growth in the annual production cycle. Analyses of variance indicated no differences among treatments (P .05), with respect to shoot height on any given date or rate of growth during the interim periods, within a vegetation type.

| Vegetation type              | Date          | Growth period (days) | Winter burn |                                   | Spring burn |                                   | Non-use     |                                   |
|------------------------------|---------------|----------------------|-------------|-----------------------------------|-------------|-----------------------------------|-------------|-----------------------------------|
|                              |               |                      | Height (cm) | Growth rate (cm d <sup>-1</sup> ) | Height (cm) | Growth rate (cm d <sup>-1</sup> ) | Height (cm) | Growth rate (cm d <sup>-1</sup> ) |
| <u>Scirpus acutus</u>        | 19 April 1981 | —                    | 18 (6)*     | —                                 | 16 (8)      | —                                 | 16 (7)      | —                                 |
|                              | 18 May 1981   | 29                   | 77 (36)     | 2.3                               | 72 (31)     | 1.9                               | 73 (28)     | 2.0                               |
|                              | 12 June 1981  | 25                   | 203 (69)    | 5.4                               | 191 (63)    | 4.8                               | 187 (57)    | 4.6                               |
|                              | 26 July 1981  | 44                   | 257 (74)    | 1.3                               | 250 (80)    | 1.4                               | 248 (66)    | 1.4                               |
|                              | 18 Aug 1981   | 23                   | 263 (62)    | 0.3                               | 260 (71)    | 0.4                               | 252 (63)    | 0.2                               |
|                              |               |                      |             |                                   |             |                                   |             |                                   |
| <u>Sparganium eurycarpum</u> | 20 April 1981 | —                    | 9 (7)       | —                                 | 8 (5)       | —                                 | 7 (7)       | —                                 |
|                              | 19 May 1981   | 29                   | 22 (10)     | 0.4                               | 18 (6)      | 0.3                               | 20 (7)      | 0.4                               |
|                              | 23 June 1981  | 35                   | 81 (19)     | 1.7                               | 70 (21)     | 1.6                               | 70 (26)     | 1.4                               |
|                              | 27 July 1981  | 34                   | 104 (26)    | 0.7                               | 93 (28)     | 0.7                               | 91 (33)     | 0.6                               |
|                              | 19 Aug 1981   | 23                   | 102 (22)    | 0                                 | 97 (25)     | 0.2                               | 94 (32)     | 0.1                               |
|                              |               |                      |             |                                   |             |                                   |             |                                   |
| <u>Carex atherodes</u>       | 21 April 1981 | —                    | 5 (4)       | —                                 | 7 (8)       | —                                 | 9 (6)       | —                                 |
|                              | 20 May 1981   | 29                   | 15 (8)      | 0.3                               | 21 (7)      | 0.5                               | 13 (5)      | 0.2                               |
|                              | 24 June 1981  | 35                   | 82 (26)     | 1.9                               | 83 (30)     | 1.8                               | 70 (22)     | 1.6                               |
|                              | 28 July 1981  | 34                   | 136 (37)    | 1.6                               | 152 (43)    | 2.0                               | 128 (31)    | 1.7                               |
|                              | 20 Aug 1981   | 23                   | 140 (32)    | 0.2                               | 148 (41)    | 0                                 | 126 (30)    | 0                                 |
|                              |               |                      |             |                                   |             |                                   |             |                                   |
| <u>Juncus balticus</u>       |               |                      |             |                                   |             |                                   |             |                                   |
|                              |               |                      |             |                                   |             |                                   |             |                                   |
|                              |               |                      |             |                                   |             |                                   |             |                                   |
|                              |               |                      |             |                                   |             |                                   |             |                                   |
|                              |               |                      |             |                                   |             |                                   |             |                                   |
|                              |               |                      |             |                                   |             |                                   |             |                                   |
| <u>Eleocharis palustris</u>  |               |                      |             |                                   |             |                                   |             |                                   |
|                              |               |                      |             |                                   |             |                                   |             |                                   |
|                              |               |                      |             |                                   |             |                                   |             |                                   |
|                              |               |                      |             |                                   |             |                                   |             |                                   |
|                              |               |                      |             |                                   |             |                                   |             |                                   |
|                              |               |                      |             |                                   |             |                                   |             |                                   |

\* All shoot height values = mean ± (standard deviation)

Fig. III.2. (a) Scirpus acutus community. (b) Sparganium eurycarpum community. (c) Carex atherodes. Shoot density and shoot weight of wetland emergent macrophytes for 2 yr following winter (W) and spring (S) burning and non-use (N) treatments. Within a community type, treatment responses with the same letter superscript were not different ( $P > .05$ ).

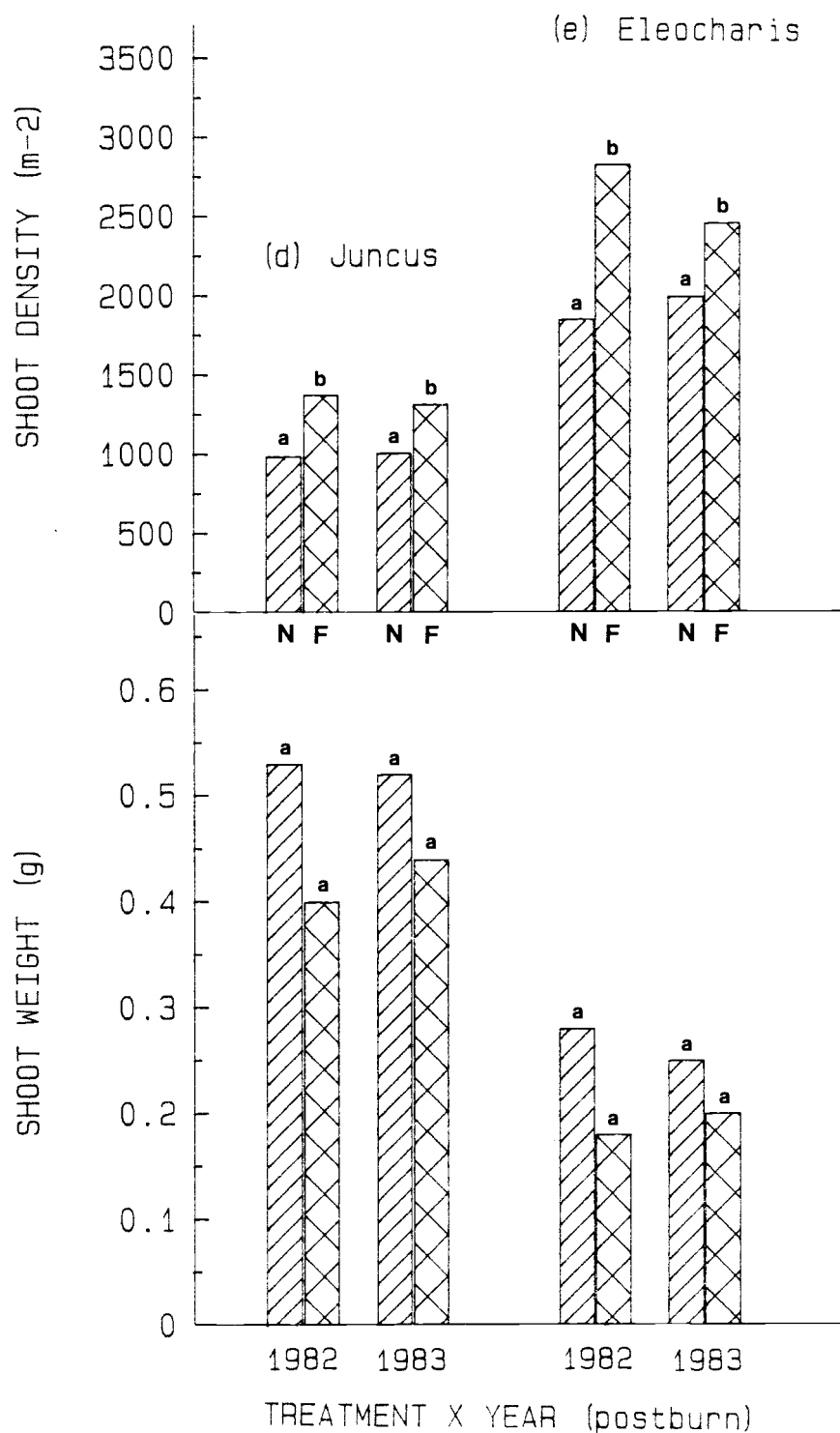


Fig. III.2



Fig. III.2. (d) Juncus balticus community. (e) Eleocharis palustris community. (see legend to Fig. III.2(a)).

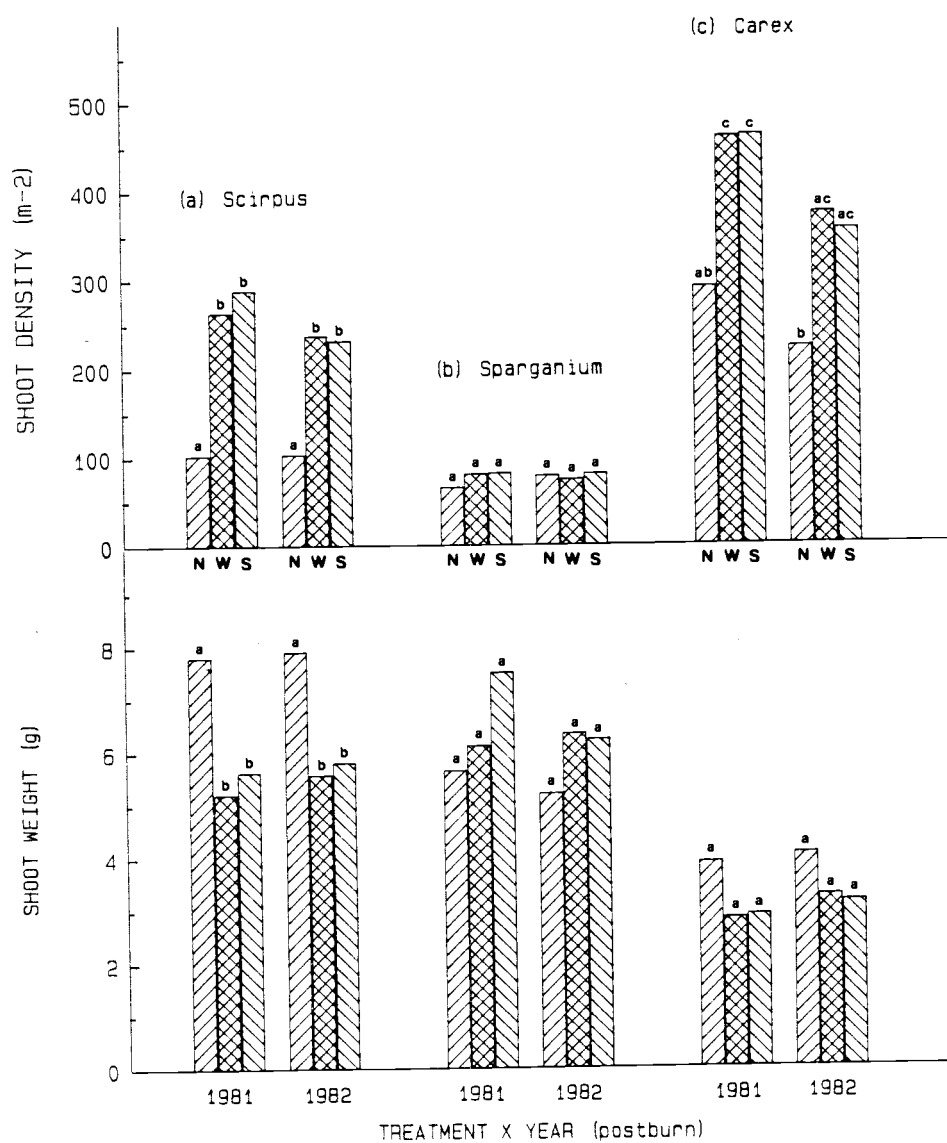


Figure III.2

Juncus and Eleocharis displayed similar, albeit nonsignificant differences ( $P > .05$ : non-use > burn shoot weight).

#### Reproductive Effort

Treatment effect on reproductive effort, measured as the percentage of shoots producing an inflorescence, was highly variable among the five wetland communities (Table III.3). Treatment had no effect on Juncus or Eleocharis vegetation, in that all shoots of these species produced inflorescences. Similarly, treatments had no effect on Sparganium reproductive effort. However, the proportion of flowering shoots in this community was much lower ( $\leq 10\%$ ) than in Juncus or Eleocharis vegetation.

Both winter and spring burning increased reproductive effort in Scirpus and Carex communities. However, this increase was only observed for one year and by the second year reproductive effort was similar among burned and non-use areas (Table III.3). Although both Scirpus and Carex increase reproductive effort after fire, their apparent similarity in response may be less meaningful when one considers the difference in total percent of flowering shoots between these two vegetation types (Scirpus  $> 40\%$ ; Carex  $< 15\%$  (Table III.3)).

Table III.3. Proportion of shoots that produced inflorescences in wetland plant communities following seasonal burning and non-use treatments. Analysis of variance was conducted using untransformed percentage data. Within a vegetation type, treatment response values followed by the same letter were not different ( $P>.05$ ).

| Vegetation type              | Year | Percentage of reproductive shoots |             |           |           |
|------------------------------|------|-----------------------------------|-------------|-----------|-----------|
|                              |      | Winter burn                       | Spring burn | Non-use   | Fall burn |
| <u>Scirpus acutus</u>        | 1981 | 68 (8)a                           | 70 (11)a    | 40 (13)b  | --        |
|                              | 1982 | 56 (9)ab                          | 61 (15)ab   | 48 (14)ab | --        |
| <u>Sparganium eurycarpum</u> | 1981 | 5 (5)a                            | 7 (6)a      | 5 (4)a    | --        |
|                              | 1982 | 6 (5)a                            | 10 (8)a     | 7 (6)a    | --        |
| <u>Carex atherodes</u>       | 1981 | 8 (3)a                            | 13 (6)a     | 2 (3)b    | --        |
|                              | 1982 | 2 (2)b                            | 1 (2)b      | 1 (2)b    | --        |
| <u>Juncus balticus</u>       | 1982 | --                                | --          | 100 (0)a  | 100 (0)a  |
|                              | 1983 | --                                | --          | 100 (0)a  | 100 (0)a  |
| <u>Eleocharis palustris</u>  | 1982 | --                                | --          | 100 (0)a  | 100 (0)a  |
|                              | 1983 | --                                | --          | 100 (0)a  | 100 (0)a  |

## DISCUSSION

### Shoot Development

Initiation of shoot growth was unaffected by burning, in any of the five wetland communities. Although this observation may, in part, be an artifact of the 10-14 day sampling interval, treatment differences would still have been minor. More importantly, any real differences in the time at which growth began are overshadowed by the similarity in shoot height when first measured and with respect to subsequent growth rates.

In a review of fire effects on wetland ecosystems Vogl (unpublished report to the US Fish & Wildlife Service, 1980) states fire stimulates increased rates of growth and sizes of vegetative plant parts. This is supported by observations at a Florida inland wetland (Vogl 1973) where Panicum hemitomon Schult. (maidencane) initiated growth earlier, increased height growth more rapidly and ultimately, achieved greater height in the first year after burning, compared to plants on an adjacent unburned site. Mook and Toorn (1982), however, found spring burning of Phragmites australis retarded initial growth of leaves and shoots for c. 1-2 weeks. Despite this, the relative growth rates and maximum heights obtained were not significantly lower than in the unburned situation. In a Great Salt Lake marsh (Smith & Kadlec 1985) growth rates, based upon biomass accumulation, were generally similar for burned and unburned Scirpus lacustris (Muhl.) Koyama (softstem bulrush), Scirpus maritimus L.

(American threesquare) and Typha latifolia. Only late in the growing season did the growth rate of unburned Typha exceed burned shoots.

Early initiation of growth after burning terrestrial grasslands has been attributed to higher soil temperatures (Daubenmire 1968). In wetlands, however, this differential response may be moderated or totally eliminated due to inundation and the overwhelming effect of insulation from solar heating by floodwaters.

#### ASC and the Dynamics of Residual Organic Materials

Dormant season burning increased ASC in Scirpus acutus, Carex atherodes, Juncus balticus and Eleocharis palustris wetlands. ASC of Sparganium eurycarpum was increased by spring burning; whereas the winter burn had no effect on production. Fuel reduction by this burn was lowest (76%) of the burns conducted; although the winter burn in Carex stands reduced fuels only slightly more (78%). Both the Sparganium and Carex winter burns were conducted when weather was cold and humid (Table III.1). Additionally, fuels were lodged over large portions of these burn units, and surface coverage by snow was greater than at sites where shoots remained upright (personal observation). These conditions account for the lower proportional fuels reduction.

Spring burns conducted over a period of four years produced a variable effect on Phragmites australis in polders of The Netherlands (Toorn & Mook 1982). Burning prior to shoot emergence had affected subsequent production only slightly. Burning after emergence,

however, severely reduced Phragmites production due to high shoot mortality. Standing crop of Scirpus lacustris and Scirpus maritimus after a September fire in a Great Salt Lake marsh was similar to production of these species in unburned areas (Smith & Kadlec 1985). Aboveground standing crop of Typha latifolia was reduced by burning, but only late in the growing season. In contrast, winter burning of Mississippi Gulf Coast marshes increased production of emergent macrophytes, including Juncus roemerianus L. (black needle rush) and Spartina spp. (cordgrasses) (Cruz & Hackney 1980). Oefinger & Scifres (1977) found production of Spartina spartinae increased on Texas coastal prairies following burning in fall, winter, or spring. A moderate intensity, January burn at a Florida inland wetland dramatically increased production of the dominant plant species Panicum hemitomon (Vogl 1973). Yields of this species were 50% and 2739% greater in standing water and adjacent wet mesic habitats, respectively, in the year after burning.

Thus it appears, burning wetlands during dormant periods will likely have little effect, or possibly, increase production of aquatic emergent macrophytes. Several authors have attributed measured or apparent gains in ASC to the reduction of heavy accumulations of residual organic materials (Lynch 1941; Ward 1968; Vogl 1973; Wade et al. 1980). In the previous reference, Vogl (1973) considered the most important vegetation change attributable to the fire to be the removal of the heavy litter mat, and attributed the increase in production largely to this factor. A similar relationship has been documented in

tall prairie vegetation (Ehrenreich 1959; Hulbert 1969) for which production is negatively correlated to litter buildup; and, when abundant, removal of litter either by fire or clipping increased production. Accordingly, the marginal (nonsignificant) increase in Sparganium ASC following the winter burn may be attributed to relatively lower fuels reduction. However, this appears inconsistent with the much greater (significant) and sustained increase in Carex ASC at the winter burn site.

The question arises as to the longevity of any increased production following burning in wetlands. Research has as yet, not evaluated the postburn dynamics of standing crop production-residual debris relationships. In this study ASC was maintained at levels greater than non-use areas for two years after burning in all but Sparganium wetlands. Significant increases of residues were measured in all communities in the second year after treatment. The buildup of litter and standing dead did not result in concurrent decline of ASC. If the rates of residue accumulation were held constant, however, preburn levels would be regained within three to five years of the burns, at which time production would also be expected to have dropped to levels comparable with non-use areas. This projection is supported by results from the Sparganium community where relatively low fuel reductions from light intensity fires are correlated with small or insignificant increases in production; and, no differences between burn and non-use by the second year. Two years after winter



burning, the amount of residual materials in Sparganium were not different from non-use.

#### Shoot Density and Shoot Weight

Increases in ASC were realized as a result of concomitant increases in shoot densities in Scirpus, Carex, Juncus and Eleocharis communities (Fig.III.1 and Fig.III.2, (a), (c)-(e)). Conversely, Sparganium ASC and shoot density were similar to non-use after burning (Fig.III.1 and Fig.III.2, (b)). Shoot weight of Scirpus, alone, was reduced by burning; although Carex, Juncus and Eleocharis shoots displayed similar (nonsignificant) trends in individual shoot biomass production.

Cruz & Hackney (1980) observed that production of a Juncus roemerianus community increased after fire, whereas, individual shoot mass was reduced. This relationship held for three years after burning. Smith & Kadlec (1985) measured species-specific responses in shoot weight after fire: Scirpus lacustris, burn<unburned; Scirpus maritimus, variable and no difference between treatments; and, Typha latifolia, burn>unburned. Spring (dormant season) burning followed by summer flooding resulted in a greater density of smaller Phragmites australis shoots (Toorn & Mook 1982). Vogl (1973) found shoot density of Panicum hemitomon increased after fire.

The variable response of this and previous studies suggests postburn shoot weight is a function of species-specific growth characteristics, fire behavior and/or postburn environmental

conditions: a complex of factors not readily controlled when investigating causative agents. The combination of greater postburn shoot densities and the trend indicating reduced individual shoot weights, in this study, suggest a density-dependent effect on biomass (Silvertown 1982). That is, individual shoot biomass is determined by the carrying capacity of a given site, which in turn is moderated by influences such as fire; and, the number of shoots emerging and surviving through the course of a growing period. This hypothesis is supported by numerous investigations of plant density-weight relationships, and is a product of self-thinning in populations of shoots (Harper 1977; Silvertown 1982).

Increases in community-wide productivity following burning are readily attributable to changes in substrate and air temperatures, solar irradiance, and nutrient availability, due to combustion of plant residues (Daubenmire 1968). In turn, improved resource levels, or rates of availability in the postburn environment may result in increased numbers of shoots, per unit area, in the early phases of growth and development (White 1981); but thereafter, inter- and/or intraspecific interactions determine equilibrium population sizes. For example, fertilization initially increases the number of ramets in a population. However, subsequent higher rates of self-thinning result in densities controlled by carrying capacity of that habitat (White 1981).

Although light, water and mineral nutrients are probably the three resources most commonly involved in determining population

interactions, the interaction among resource factors makes it impossible to isolate the partial effect of any one resource on site productivity, much less on the parameters shoot density and weight (Harper 1977). The addition of fire as an environmental influence only serves to complicate these relationships.

Increases in shoot density appear to be a consistent response of rhizomatous emergent macrophytes, under conditions of dormant season, surface fires. In this situation, regenerative tissues arising from the perennial rhizomes are protected from heat injury. Peat or ground fires, however, consume all or portions of organic substrates, the zone where most rhizomes of emergent species occur (Lynch 1941; Hoffpauer 1968). Peat fires may severely reduce or eliminate emergent vegetation, producing areas of open water and initiating a successional sequence with aquatic submergent plants filling the role of pioneer species. Thus, fire intensity again seems to be an important variable affecting postburn community structure and function.

#### Reproductive Effort

Reproduction (sexual) and growth (vegetative) are alternative means by which a plant may use limited resources. The relative allocation of such resources affects the fitness, or relative evolutionary advantage of an individual (Silvertown 1982). Reproductive effort has been used to measure resource allocation in organisms (MacArthur & Wilson 1967). In this study reproductive effort is used to test the assumption that fire illicit a change in

the pattern of allocating resources. Reproductive effort has been defined on the basis of seed production and biomass (Thompson & Stewart 1981). Both approaches are subject to criticism, however, in that seed output does not measure allocation to all reproductive structures. Furthermore, in many species reproductive structures contain chlorophyll and after an initial growth stage can significantly contribute or fully supply photosynthates for the energy cost of their production (Bazzaz & Carlson 1979). Both methods are further complicated by the difficulty of determining the appropriate sampling period. To avoid these problems we used the simpler approach taken by other workers (Thompson & Stewart 1981), treating reproductive effort as the proportion of a population of shoots that produce inflorescences.

Reproductive effort varied among emergent macrophytes at Malheur NWR and, furthermore, displayed equally variable responses to fire. Percentage of reproductive shoots in non-use areas ranged from zero in localized patches of Sparganium and Carex, to 100% in Juncus and Eleocharis communities. The large allocation for seed production in Eleocharis is likely related to its seral status as it is frequently first to occupy recently disturbed sites in medium to shallow waters (Lynch 1941; Hoffpauer 1968; Cronquist et al. 1977; personal observation). The high level of reproductive effort of Juncus indicates seed production may be equally important in the sense of long-term population maintenance in a variable environment (Harper 1977).

Only Scirpus and Carex increased reproductive effort following burning; and even then, differences relative to non-use were shortlived. The absolute increases observed illustrate species-specificity of these vegetation types: Scirpus inflorescence production increased by >25% in the first year after burning; Carex seedhead production following fire increased on the average <10%. Relative increases in reproductive effort, however, were quite different at c. 15-25% and 400-600% for Scirpus and Carex, respectively.

Vogl (unpublished report submitted to the US Fish & Wildlife Service, 1980) contends fire in wetlands stimulates production of floral components, fruit and seeds. Prior observations, although largely unquantified, support this position. Carex oligosperma (few-seeded sedge) and Eriophorum spissum (cotton grass) displayed "conspicuous" seed production after burning of northern Wisconsin sedge meadows (Vogl 1964). Additionally, flowering activity of Polygonum hydropiperoides, measured as percent frequency of plants in flower, was much greater in burned portions of a Florida wetland (Vogl 1973). Other research apparently, has not considered fire effects on reproductive allocation in aquatic emergent macrophytes.

Improved resource availability, especially of the mineral nutrients nitrogen and phosphorous, increases total reproductive output in numerous herbaceous species (Harper 1977); and, this is the mechanism attributed to increased flowering responses following fire (Daubenmire 1968; Volland & Dell 1981). Results of this study suggest

reproductive effort of emergent species is variable and specific to individual species, within the context of burning conditions. However, plants with strong vegetative growth habits may be less dependent on sexual reproductive strategies and, therefore, less plastic in response to environmental influences such as fire.

SECTION IV

FIRE EFFECTS ON UPLAND VEGETATION OF  
MALHEUR NATIONAL WILDLIFE REFUGE  
SOUTHEASTERN OREGON

Richard P. Young

Richard F. Miller

FIRE EFFECTS ON UPLAND VEGETATION OF  
MALHEUR NATIONAL WILDLIFE REFUGE  
SOUTHEASTERN OREGON

SUMMARY

(1) Vegetation response to dormant season fires (winter, early spring) and non-use were investigated in upland habitats occupied by the plant communities: Chrysothamnus nauseosus/Elymus cinereus shrub-grassland; Elymus triticoides mesic grassland; and, Sarcobatus vermiculatus/Distichlis spicata shrub-grassland. These are terrestrial bottomland habitats of a wetland-upland complex, and are transitional from aquatic emergent macrophyte communities to terrestrial sites occupied by the regional, semiarid shrub-grass vegetation.

(2) Individual Chrysothamnus spp. and Sarcobatus shrubs were monitored for one to two seasons after burning. All shrubs were top-killed, but mortality was  $\leq 5\%$  for any burn. Regrowth from basal sprouts was rapid, with canopy coverages and canopy volumes attaining 40-60% of preburn sizes, two years after treatment. Few seedlings were observed after the fires, and none survived a single growing season in non-use or burn areas.

(3) Fire effects were observed for Elymus cinereus bunchgrasses sampled before and after burning. Reductions of live basal (=crown) area due to heat injury were offset by increased basal growth in other individuals. Shoot density, herbage production, and the proportion of



reproductive shoots increased for two years after burning. No recruitment from seedling establishment was observed in non-use or burn areas during this period.

(4) Aboveground standing crop (ASC) increased for one to two years following burning, in all vegetation types. Increased production was due principally to the positive response of perennial graminoids. Other classes of herbaceous species displayed similar trends in ASC, however, treatment and year differences were seldom significant ( $P > .05$ ) due to high sample variability.

(5) Shoot density of the rhizomatous graminoids Elymus triticoides, Carex douglasii and Juncus balticus increased for one year after burning, but were not different from non-use areas by the second growing season. Similar trends were observed for Elymus and Carex with respect to the proportion of reproductive shoots.

(6) Vegetation response to dormant season burns was dramatic, but short-lived. Changes were quantitative rather than qualitative, with little or no change in composition. Some implications of these findings are discussed, including a consideration of the historical role of fire in these habitats.

## INTRODUCTION

Fire is a natural component of inland wetlands and associated terrestrial (syn.: upland) ecosystems (Viosca 1928, 1931; Lynch 1941; Vogl 1969; Davies 1979). Additionally, fire is frequently used as a management technique in these habitats (Schlichtemeier 1967; Vogl 1967; Ward 1968; Wade et al. 1980). Although the use of fire in ecosystem management depends largely upon a knowledge of fire effects on vegetation (Wright & Bailey 1982), few studies have considered the role of fire in inland wetlands (Vogl 1980). Marshes of semiarid regions of the western United States have only recently been investigated (Smith & Kadlec 1985), and we have found no references to fire effects in adjacent upland communities, where these differ from the major regional vegetation. Thus there is a need to better understand the influence of fire on the structure and functioning of these ecosystems.

We studied vegetation response to fire in a wetland-upland complex. This paper reports findings from the terrestrial habitats we evaluated. Details of the wetland studies are given in a separate paper (Section III).

### Approach

Plant response to environmental influences such as fire may best be understood when approached from more than a single level of hierarchy or integration (Young 1983; Young & Miller 1985). Thus we chose to study selected vegetation attributes at the individual plant,

population and/or community scale. However, all parameters were analyzed within the context of two general hypotheses: (1) fire has no effect on the vegetation parameter of interest, relative to a non-use condition, i.e. no burning, livestock grazing, mowing, etc.; and, (2) vegetation response is independent of the season of burning.

#### STUDY AREA

Field experiments were conducted at Malheur National Wildlife Refuge (NWR) in southeastern Oregon. Malheur lies within internally drained Harney Basin (elevation: 1250-1280 m), which forms the northwestern portion of the Basin and Range Province of the western United States (Baldwin 1964). It consists primarily of flat to gently rolling lowlands derived from pluvial lake sediments and alluvium (Walker & Swanson 1968). Cold desert-shrub vegetation of the upland sites, dominated by species of the Asteraceae and Chenopodiaceae families, reflect the regional semiarid climate (Franklin & Dyrness 1973).

Typical of many high elevation basins of the Intermountain region, Malheur is characterized by a semiarid climate and moderate to cold temperatures. Annual precipitation for the basin averages 25-30 cm, but extremes of 10 cm and 50 cm have been recorded (Gomm 1979). The bulk of precipitation occurs in winter and spring, whereas summers are typically dry. Growing seasons are limited by cool spring temperatures, dry summers, and freezing temperatures which generally occur in late August to early September (Gomm 1979).

Extensive areas of wetland vegetation occur in and around the terminal lakes and throughout the riverine and palustrine zones of watercourses entering the basin. Lowland habitats, exclusive of the terminal lakes, consist of a diverse association of wetland and terrestrial ecosystems. Wetlands form a continuum of ponds, sloughs, marshes and wet meadows. Adjacent terrestrial communities finger into the wetlands, frequently forming islands of upland vegetation. Thus wetland-upland vegetation of these areas constitute a complex ecosystem with a mosaic pattern of plant communities and open water.

Dominant emergent plant species of wetland communities include Scirpus acutus Muhl. (hardstem bulrush), S. americanus Pers. (American threesquare), S. maritimus L. (alkali bulrush), Typha latifolia L. (common cattail), Sparganium eurycarpum Engelm. (bur-reed), Carex atherodes Spreng. (awned sedge), C. nebraskensis Dewy (Nebraska sedge), Juncus palustris (L.) Roemer & Schultes (spikerush) (Deubbert 1969; Copeland 1979). In wet meadows and small basins flooded for only short periods in the spring, and in the upland mesic meadows, emergent species give way to the true grasses. The most important of these at Malheur NWR are Distichlis spicata var. stricta (L.) Greene (inland saltgrass), Elymus triticoides Buckley (creeping wildrye), and Poa nevadensis Vasey ex. Scribn. (Nevada bluegrass).

Terrestrial habitats of the wetland-upland complex display affinities to both the more mesic wetlands and to the more xeric plant communities of elevated landscapes. Plant species of both floras combine in transitional communities with representatives from

emergent and terrestrial habitats that exchange dominance as the effective environment dictates. Species contributed from surrounding uplands are typical of the Atriplex confertifolia (shadscale) and Artemisia tridentata Nutt. (sagebrush) associations (Franklin & Dyrness 1972). Common shrub species of lowland, terrestrial sites include Artemisia tridentata, Chrysothamnus nauseosus var. albicaulis (Nutt.) Rydb. (white-stem rabbitbrush) Chrysothamnus viscidiflorus var. viscidiflorus (Hook.) Nutt. (green rabbitbrush), and Sarcobatus vermiculatus (Hook.) Torr. (greasewood). Important grasses include the perennial bunchgrass Elymus cinereus Scribn. & Merr. (Basin wildrye) and the exotic annual species Bromus tectorum L. (cheatgrass).

The upland sites selected for study include the following:

1. Chrysothamnus nauseosus/Elymus cinereus (syn.: CHNA/ELCI) community. This is a mixed shrub-grass community with C. nauseosus and Sarcobatus sharing dominance, and C. viscidiflorus the subdominant of the shrub union. Dominant herbaceous species frequently vary across short distances, but typically include Elymus cinereus, E. triticoides or Bromus tectorum (Appendix F). These sites occur as elevated strands on dune-like landforms. Soil textures are sandy loams and loamy sands (field estimate), from the surface to 1.3+ m depth. These sites occupy a position of greatest elevation above summer floodwater levels. Ease of root penetration and high plant-available water holding capacity, characteristic of these soil textures (Buckman & Brady 1969), probably result in a habitat more mesic than indicated merely by elevation above adjacent wetlands. Of

primary concern in this community were the responses to fire of the dominant/subdominant shrubs and grass species, at the population and individual plant levels; and, community-level responses of the functional groups -- perennial grasses, perennial forbs, annual grasses and annual forbs.

2. Elymus triticoides (syn.: ELTR.) community. This is an herbaceous association dominated by the rhizomatous graminoids (in decreasing order of importance) E. triticoides, Carex douglasii and Juncus balticus (Appendix E). A diverse mix of other graminoids and forbs is common. These habitats are nearly level. Soils are derived from alluvial sediments of primarily very fine sand, silt and clay soil particle fractions, with silt loam and silty clay loam textures (field estimate) from the soil surface to a depth of 1.0+ m. Depth to free water was <15 cm in May, and exceeded 1.0 m in October of both postburn sample years (personal observation). Primary interest in this community centered on the response of the three dominant graminoids, at the population and individual plant (=shoot) levels, and; on the community-level response of the annual and perennial groups of graminoids and forbs.

3. Sarcobatus vermiculatus/Distichlis spicata (syn.: SAVE/DISP) community. This is a shrub-grass mixture with Sarcobatus and Distichlis the sole dominants of their respective unions (Appendix G). Other species are minor components of the stand. The site occurs on Pelican Island in Malheur Lake, a terminal lake within the basin. It is nearly level and is the most mesic of the three upland habitats.

Depth to free water was never observed to be >28 cm. Observations were terminated one year after a single burn treatment, due to rising lake-levels and inundation of the study area. Soils were loamy sands and sandy loams of primarily fine and very fine sand particle fractions (field estimate) in the surface 25 cm of the profile. Principle interests at this site were individual plant and population-level responses of Sarcobatus and Distichlis, a rhizomatous species, and the community level response of the plant groups -- other perennial and all annual herbaceous species.

## METHODS AND MATERIALS

### Field Methods

Treatments included winter and early-spring, i.e. dormant season burns, and non-use in the CHNA/ELCI and ELTR habitats. Five replicate sites were located for each vegetation type-treatment combination. A single spring burn was conducted in the SAVE/DISP study area, and contrasted with an adjacent non-use area. Eight replicates (=experimental units) were located in each of these areas.

Prior to initiation of these studies, all sites had been free of cultural practices and wildfires for a minimum of three years. A minimum treatment area size of 2 ha was necessary for possible selection; however, the smallest unit included exceeded 3 ha.

To test the assumption that treatment areas within a vegetation type were comparable prior to burning, herbaceous standing crop

(current and previous year's components) and shrub canopy cover were sampled following the 1980 growing season. Herbaceous materials were harvested to ground level in 10 to 20 quadrats 0.5 x 0.5 m, using appropriate randomization procedures (Steel & Torrie 1980); 32 to 40 transect lines and belts (1.0 m width) were similarly located to determine line-intercept shrub canopy cover and shrub density (see Appendices B, C, D, F and G for details of sampling and summary of findings).

#### Controlled Burning

Fuel load of fine fuels was determined in each study area shortly before the burns were conducted. Harvest sample methods, similar to those described above, were used. Immediately prior to burning, fine fuels were sampled for fuel moisture determination (N=10-20 per treatment; oven-dry @ 60 C for 48 hr; dry weight based moisture content). temperature, relative humidity and wind speed were measured periodically during each burn.

Treatment areas were ignited using strip headfires (Martin & Dell 1978). Following each burn fuels were resampled to determine the amount of fuels combusted. These data were used to determine fire intensity, the rate of energy release per unit time per unit length of fire front (Byram 1959). Fire intensity is a useful measure for evaluating fire effects and comparisons among various burns (Rothermel & Deeming 1980). Calculation of fire intensity followed the procedures outlined by Alexander (1982), using a basic value of



Table IV.1. Fuels, weather and fire behavior for experimental prescribed fires conducted in upland habitats at Malheur NWR, 9 December 1980 to 15 April 1981. ---: no shrubs present.

| Habitat   | Burndate      | Preburn fire<br>fuel load<br>(g m <sup>-2</sup> ) | Preburn<br>shrub<br>cover (%) | Fuel<br>moisture<br>(%) | Temper-<br>ature<br>(C) | Relative<br>humidity<br>(%) | Wind<br>speed<br>(km h <sup>-1</sup> ) | Headfire<br>intensity<br>(kW m <sup>-1</sup> ) | Fine fuels<br>reduction<br>(%) |
|---|---------------|---|-------------------------------|-------------------------|-------------------------|-----------------------------|--|--|--------------------------------|
| <u>Chrysothamnus/</u><br><u>Elymus cinereus</u> | 9 Dec 1980    | 486 (43-2502)                                     | 14.2                          | 16.3                    | 3-5                     | 27-34                       | 2-16                                   | 260-17862                                      | 63                             |
|   | 7 April 1981  | 461 (72-1485)                                     | 12.7                          | 8.2                     | 4-9                     | 31-35                       | 6-10                                   | 260-23879                                      | 68                             |
| <u>Elymus triticoides</u>                       | 10 Dec 1980   | 679 (464-839)                                     | ---                           | 15.5                    | 2-3                     | 47-53                       | 2-13                                   | 1888-3431                                      | 81                             |
|   | 7 April 1981  | 586 (398-944)                                     | ---                           | 6.3                     | 6-8                     | 29-38                       | 10-19                                  | 3721-8826                                      | 90                             |
| <u>Sarcobatus/</u><br><u>Distichtis</u>         | 15 April 1981 | 968 (722-1470)                                    | 12.5                          | 7.0                     | 20-22                   | 26-33                       | 13-23                                  | 13411-27306                                    | 94                             |

\* Mean (range)

18700 kJ kg<sup>-1</sup> for low heat of combustion. A summary of fuels, weather and fire behavior for each burn is given in Table IV.1.

The burns were generally of low to moderate intensity (McArthur & Cheney 1966). Flaming fronts moved steadily through the accumulations of fine fuels. Fuels consumption was >60% in all cases, with the lowest and most variable reductions in the CHNA/ELCI study area.

#### CHNA/ELCI Vegetation Sampling

Pretreatment sampling included the standing crop (Appendix C) and shrub cover (Appendix F) measurements previously described and estimates of herbaceous species frequency (200 quadrats, 0.5 x 0.5 m, per treatment, Appendix F). Sampling intensity of standing crop measurements was too low for comparison with postburn measurements. Additionally, frequency sampling proved insensitive to the fire-induced community changes we observed. These data are provided, therefore, for the purpose of community characterization and especially, as a measure of species abundance within each community.

Four shrubs of Chrysothamnus nauseosus and Sarcobatus vermiculatus and two of C. viscidiflorus were randomly selected in each replicate (N=20, 20 & 10 shrubs per treatment, respectively). Following each growing season shrub canopies were measured for determination of horizontal canopy coverage and canopy volume (absolute sizes) following the methods of Phillips & MacMahon (1981). Canopy development was then calculated with respect to pretreatment shrub size.

Community-wide canopy cover of these species was obtained by line-intercept: four transects, each of length 10 m, per replicate (N=20 transects per treatment; Appendix F). Shrub seedling establishment was monitored in 1 m wide belts adjacent to these transects. The entire belt or strip-transect was carefully searched for new seedlings in September of 1981 and 1982.

Individual Elymus cinereus plants were similarly located and marked prior to burning: four plants per replicate (N=20 plants per treatment). Pretreatment measurements involved mapping of plant basal areas and determination of the live basal (=crown area (Young & Miller 1985)). In 1981 and 1982 plant basal areas were remeasured, permitting us to evaluate fire-induced changes for this parameter. At the same time measurements were taken in three microquadrats, 5 x 10 cm, systematically located (approx. equidistant around the basal perimeter) within the crown of each plant, including: (1) a count of live shoots to estimate shoot density per unit live basal area; (2) a count of all shoots that produced an inflorescence to determine the proportion of reproductive shoots; and, (3) harvest all live shoots to ground level to determine ASC per unit live basal area.

Herbage production was estimated by a single harvest of ASC of current year's materials and residual materials (=litter). Sampling was conducted in late July of 1981 and 1982. Observations in 1980, 1981 and 1982 suggest this period was near that of peak ASC, in that maximum shoot height had been achieved and emergence of new shoots had ceased (personal observation). All aboveground materials were

collected in 0.5 x 0.5 m quadrats; 12 quadrats per replicate (N=60 quadrats per treatment). Harvested materials were separated into vegetation components based upon life form, and litter.

#### ELTR Vegetation Sampling

Pretreatment ASC and frequency sampling were conducted as in the CHNA/ELCI community (Appendix B, E). As previously, these measurements were used to test for homogeneity among treatment study sites (Appendix B), and to provide a description of species abundance (frequency of occurrence) in these vegetation communities (Appendix E).

Herbage production was similarly estimated from harvest samples obtained in early August of 1981 and 1982. Sample size was eight quadrats per replicate (N=40 quadrats per treatment). At this time, shoot density and numbers of reproductive shoots were determined in 0.01 m<sup>2</sup> microquadrats for E. triticoides, Carex douglasii and Juncus balticus.

#### SAVE/DISP Vegetation Sampling

Pretreatment shrub cover and ASC measurements were obtained in a manner similar to those described above. Individual Sacobatus shrubs were located prior to burning, marked, and subsequently canopy development was monitored as in the CHNA/ELCI communities.

Herbage production was estimated, as before, from harvests of ASC and litter. Sample size was four quadrats per replicate (N=32 quadrats per treatment). Measurements were obtained for one year after the single spring burn (1981), as rising lake levels inundated

these areas beginning in winter 1981-82. Observations in this habitat were restricted due to the island location and access difficulties.

### Analysis

Pretreatment ASC measurements were used to determine adequate sample sizes for postburn herbage production sampling. The preliminary sampling provided a variance estimate that was used in Stein's two-stage procedure. Sample sizes were projected to provide estimates with  $\pm 10\%$  of the true population mean in 90% of all cases (Steel & Torrie 1980), for the vegetation component = perennial graminoids. High variability precluded obtaining this degree of accuracy/precision for the less abundant components.

Analyses of variance for factorial experiments (Cochran & Cox 1966) were used to test for treatment (burned, non-use), time (years after treatment) and treatment x time effects for the plant parameters discussed above. Quadrats and transects sampled within each replicate were treated as subsamples in these analyses. All F tests were conducted using the experimental error term, from which the sampling error had been partitioned. Where treatment effects were significant ( $P < .05$ ), least significant difference (lsd) tests were conducted for all possible paired comparisons (Steel & Torrie 1980).

### RESULTS

Spring growth began in mid- to late April in 1981 and 1982, in the CHNA/ELCI and ELTR communities (access precluded regular visits to

the SAVE/DISP study area). Although green-up was initiated on burn areas before non-use, differences in timing were minor (<3 to 5 days) and appeared negligible after two to three weeks of growth. Flowering activity of individual species was temporally indistinguishable in either 1981 or 1982.

#### Shrub Species Response

All marked shrubs in the CHNA/ELCI and SAVE/DISP areas were top-killed by their respective burns. Two Sarcobatus shrubs from the SAVE/DISP spring burn were completely killed. All other marked plants vigorously resprouted in the year after the burns.

There were no differences in canopy cover or volume for non-use shrubs over the three years of measurements (Fig. IV.1). Regrowth of burned shrubs was similar among treatments and for the two posttreatment years. C. viscidiflorus was most rapid to recover, although differences among species were not significant ( $P > .05$ ). After two years of regrowth, canopy cover was 36-56% of preburn values, and canopy volume was 50-58% of preburn size: in both cases -- C. viscidiflorus > C. nauseosus > Sarcobatus. After one year of regrowth, Sarcobatus shrubs of the SAVE/DISP burn area had achieved canopy cover (28.3%) and volume (22.7%) development only slightly greater than, but not significantly different ( $P > .05$ ) from that of plants in CHNA/ELCI treatments.

Fig. IV.1. Shrub canopy development as a percentage of pretreatment (September 1980) canopy cover and canopy volume, for two years after burning. CHNA/ELCI community: (a) Chrysothamnus nauseosus (N=20); (b) Sarcobatus vermiculatus (N=20); (c) Chrysothamnus viscidiflorus (N=10). N: non-use; W: winter burn; S: spring burn.

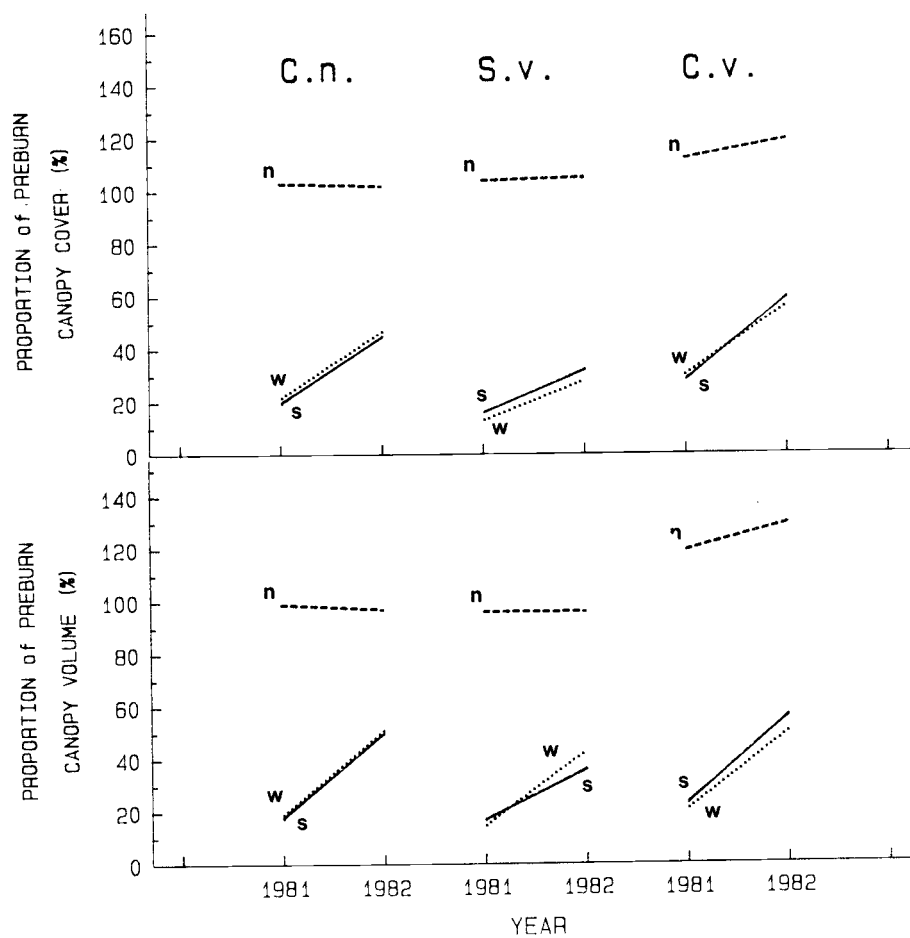


Fig. IV.1



## Herbaceous Species Response

Elymus cinereus

Live basal areas of E. cinereus plants were similar among and within treatments, relative to their 1980 sizes (Table IV.2). Despite this, individual plants of burned sites suffered reductions in basal area of nearly 30%. Many individuals, however, displayed increases in basal area (maximum increase = 16%; spring burn treatment) as a result of vigorous shoot development around the crown perimeter. Basal areas continued to expand during the second year, but differences were not significant.

Total shoot density per unit live basal area increased in the first year following winter and spring burns (Table IV.2). By the second year, however, shoot density of burned and non-use plants was not different. The percentage of reproductive shoots also increased after fire (Table IV.2). Time of burning had no effect on this attribute in 1981 or 1982. Reproductive shoots decreased from 1981 to 1982 for both burns; although second year percentages still exceeded non-use values.

ASC per unit live basal area increased after burning (Table IV.2). There was no difference among burns or years in 1981 and 1982; although the trend was a decrease in production from 1981 to 1982. Using these data we calculated mean shoot weights and found production patterns, on an individual shoot basis, mimic those of total ASC, i.e. (1) no difference among years for non-use (2.74 g and 2.52 g, 1981 and 1982, respectively), winter burns (3.40 g, 3.20 g), or spring

Table IV.2. Elymus cinereus response to non-use and winter and spring burning: (1) plant basal area as a percentage (%) of pretreatment (September 1980) size; (2) shoot density ( $\text{dm}^{-2}$ ); (3) aboveground standing crop (ASC) per unit basal area ( $\text{g dm}^{-2}$ ); (4) reproductive shoots as a percentage of total shoots (%). Values are mean (standard deviation). Within categories, values followed by the same letter are not different ( $P > .05$ ).

|                     |      | Non-use      | Winter burn   | Spring burn   |
|---------------------|------|--------------|---------------|---------------|
| Basal area          | 1981 | 99.3 (6.2)a  | 102.4 (3.4)a  | 102.5 (2.8)a  |
|                     | 1982 | 99.4 (8.1)a  | 107.0 (3.7)a  | 106.7 (3.1)a  |
| Shoot density       | 1981 | 34.6 (6.1)a  | 48.2 (5.2)b   | 49.4 (6.0)b   |
|                     | 1982 | 35.5 (5.8)a  | 40.3 (4.8)a   | 42.1 (5.5)a   |
| ASC/basal area      | 1981 | 94.7 (22.1)a | 154.0 (23.4)b | 162.1 (20.1)b |
|                     | 1982 | 89.3 (18.7)a | 137.2 (17.8)b | 131.3 (16.2)b |
| Reproductive shoots | 1981 | 17.1 (10.4)a | 77.5 (8.3)c   | 71.9 (12.0)c  |
|                     | 1982 | 13.3 (14.1)a | 53.4 (8.2)b   | 56.4 (7.9)b   |

burns (3.28 g, 3.12 g); (2) no difference due to season of burning; and, (3) increased weight per shoot for two years after the fires.

#### Herbage Production, Litter Replacement

Herbage production, as estimated by ASC, was generally consistent among the three habitats, in their response to fire. In the CHNA/ELCI community, burns had the similar effect of increasing total ASC, and ASC of Elymus cinereus and all other perennial graminoids (Table IV.3). Production of all annuals combined increased for one year following the burns. No differences are attributable to season of burns. Meaningful differences among the other vegetation components, if they do exist, are masked by the relatively high variance terms associated with sample estimates. For example, burning appears to increase annual grass (Bromus tectorum) production in the first postburn year (Table IV.3); and this trend is supported by field observations (personal observation). Despite this, spatial variation was sufficiently large that this difference is not significant ( $P > .05$ ). Litter build-up on burned sites was also subject to high variation (Table IV.3). However, treatment and year differences were also quite dramatic and by the second year, litter replacement amounted to about 1/3 to 2/3 of non-use values.

Burning in ELTR communities increased ASC of Elymus triticoides for one year (Table IV.4). Production of all other perennial graminoids and perennial forbs increased for one year; but differences were not significant despite the obvious trend in ASC values and in

Table IV.3. Aboveground standing crop (ASC) of herbaceous vegetation and litter ( $\text{g m}^{-2}$ ) for non-use and winter and spring burns in the CHNA/ELCI community. Values are mean (standard deviation) for two years (1981, 1982) following treatment. Values followed by the same letter are not different ( $P > .05$ ).

|                               | Non-use     |            | Winter Burn |             | Spring Burn |            |
|-------------------------------|-------------|------------|-------------|-------------|-------------|------------|
|                               | 1981        | 1982       | 1981        | 1982        | 1981        | 1982       |
| <i>Elymus cinereus</i>        | 177 (93)a   | 156 (87)a  | 449 (186)d  | 357 (118)d  | 316 (106)d  | 286 (124)d |
| Other perennial<br>graminoids | 81 (31)b    | 68 (30)b   | 189 (52)a   | 108 (33)a   | 146 (37)a   | 123 (40)a  |
| Perennial forbs               | 4 (7)c      | 1 (6)c     | 5 (9)c      | 2 (6)c      | 1 (5)c      | 1 (3)c     |
| Annual grasses                | 4 (14)c     | 2 (15)c    | 21 (14)c    | 8 (13)c     | 16 (15)c    | 4 (6)c     |
| Annual forbs                  | 6 (11)c     | 3 (5)c     | 10 (12)c    | 6 (4)c      | 4 (7)c      | 4 (8)c     |
| Perennials (total)            | 262 (121)ad | 223 (112)a | 643 (181)e  | 466 (154)de | 462 (134)de | 410 (152)d |
| Annuals (total)               | 10 (12)c    | 5 (13)c    | 31 (17)f    | 14 (12)cf   | 20 (16)f    | 8 (7)c     |
| Total ASC                     | 272 (135)ad | 228 (116)a | 674 (206)e  | 479 (133)d  | 482 (120)de | 417 (112)d |
| Litter                        | 346 (154)d  | 382 (148)d | 10 (22)c    | 127 (36)a   | 6 (18)c     | 206 (56)a  |

Table IV.4. Aboveground standing crop (ASC) of vegetation components and litter ( $\text{g m}^{-2}$ ) in the Elymus triticoides community; for two years (1981, 1982) following burning treatments. Values are mean (standard deviation); values followed by the same letter are not different ( $P>.05$ ).

|                               | Non-use    |            | Winter Burn |            | Spring Burn |            |
|-------------------------------|------------|------------|-------------|------------|-------------|------------|
|                               | 1981       | 1982       | 1981        | 1982       | 1981        | 1982       |
| <u>Elymus triticoides</u>     | 99 (33)d   | 156 (34)de | 217 (55)ef  | 178 (29)ef | 267 (64)f   | 195(19)ef  |
| Other perennial<br>graminoids | 52 (16)c   | 56 (20)c   | 73 (17)c    | 52 (27)c   | 82 (21)c    | 58 (24)c   |
| Perennial forbs               | 4 (6)a     | 2 (4)a     | 12 (7)ab    | 9 (8)ab    | 19 (7)b     | 13 (8)ab   |
| Annuals                       | 0          | >1         | 2 (5)       | >1         | 0           | 0          |
| Total ASC                     | 155 (42)d  | 214 (55)de | 303 (72)e   | 239 (60)de | 368 (84)e   | 266 (58)de |
| Litter                        | 733 (148)g | 548 (109)g | 75 (22)c    | 175 (61)d  | 23 (16)bc   | 216 (79)d  |

plant size/vigor as observed in the field (personal observation). Litter replacement was approximately 1/3 to 2/5 of non-use amounts after the second year.

ASC of Distichlis increased after burning in the SAVE/DISP community: burn=550  $\pm$ 125 g m<sup>-2</sup>; non-use=398 $\pm$ 74 g m<sup>-2</sup> (P<.05). Production of all other species combined accounted for 10% (60 $\pm$ 68 g m<sup>-2</sup>) and 5% (23 $\pm$ 25 g m<sup>-2</sup>), respectively, of total ASC on burn and non-use sites. Differences were not significant (P>.05).

#### Shoot Density, Reproductive Shoots: Rhizomatous Graminoids

Burning increased shoot density of Elymus triticoides for one year in both winter and spring burns (Table IV.5); and, the proportion of reproductive shoots of Elymus and Carex douglasii for one growing season. Trends indicate increased shoot density additionally for Juncus balticus, and for a second year among all three of these species. Again, however, these trends were not supported by statistical analyses (Table IV.5).

The proportion of Juncus reproductive shoots exceeded 85% in all cases. Treatment and year differences were small and not significant for this species.

## DISCUSSION

In both the CHNA/ELCI and ELTR habitats, timing of burning within the dormant season had no influence on subsequent vegetation response (Fig. IV.1, Table IV.2, IV.3, IV.4, IV.5). This pattern was similarly

Table IV.5. Total shoot density ( $\text{dm}^{-2}$ ) and percentage reproductive shoots (%) for the dominant graminoids of the Elymus triticoides community; treatments were non-use and winter and spring burns. Data are for two years (1981, 1982) after burning. Values are mean (standard deviation); values followed by the same letter are not different ( $P>.05$ ).

|                                | Non-use    |            | Winter Burn |             | Spring Burn |             |
|--------------------------------|------------|------------|-------------|-------------|-------------|-------------|
|                                | 1981       | 1982       | 1981        | 1982        | 1981        | 1982        |
| Total shoot density            |            |            |             |             |             |             |
| <u>Elymus triticoides</u>      | 3.2 (1.4)a | 3.5 (1.2)a | 6.3 (1.5)b  | 4.5 (1.4)ab | 6.7 (1.8)b  | 4.6 (1.5)ab |
| <u>Carex douglasii</u>         | 6.8 (2.1)b | 6.7 (1.7)b | 8.2 (2.4)b  | 7.0 (2.2)b  | 8.0 (1.7)b  | 7.1 (1.9)b  |
| <u>Juncus balticus</u>         | 1.3 (1.5)a | 1.5 (1.1)a | 2.4 (1.7)a  | 2.0 (2.1)a  | 2.6 (1.9)a  | 2.0 (1.3)a  |
| Percentage reproductive shoots |            |            |             |             |             |             |
| <u>Elymus triticoides</u>      | 31 (18)c   | 35 (14)c   | 57 (21)cd   | 48 (16)cd   | 63 (16)d    | 55 (19)cd   |
| <u>Carex douglasii</u>         | 22 (12)e   | 24 (10)e   | 42 (10)f    | 35 (18)ef   | 45 (11)f    | 34 (23)ef   |
| <u>Juncus balticus</u>         | 87 (15)g   | 94 (22)g   | 98 (19)g    | 95 (18)g    | 97 (13)g    | 90 (20)g    |

observed in several wetland habitats, including an emergent stand of Juncus balticus (see Section III). Despite these similarities, it is worth a cautionary note to point out the lack of replication for burn treatments over years.

#### CHNA/ELCI Community

The sprouting response of Chrysothamnus nauseosus and C. viscidiflorus, as well as that of Sarcobatus has previously been documented (Blaisdell 1953; Robertson & Cords 1957; Daubenmire 1970; Mohan 1973). However, previous reports have evaluated shrub response by means of more extensive sampling procedures such as canopy cover by line intercept or quadrat-ocular estimation. The approach we have utilized permits us to evaluate individual plant responses, and to extrapolate these to the population of shrubs. In these studies, shrub response was almost universally consistent within and among species.

Regrowth following burning was rapid with respect to replacement of canopy cover and volume (Fig. IV.1). Previous measurements for these Chrysothamnus spp. are for canopy cover only, and these indicate recovery rates of much less than to about the same as the values we observed (Blaisdell 1953; Young & Evans 1974). Lower response rates are likely, in part, a function of the more xeric environments of other studies. However, Chrysothamnus regrowth after burning may be quite rapid even on relatively harsh sites where postburn competition from



herbaceous species is reduced (McKell & Chilcote 1957; Young & Evans 1974).

Regrowth rates for Sarcobatus have not been previously reported. Our results indicate little or no mortality due to fire, and regrowth rates similar to Chrysothamnus spp.

Fire effects on Elymus cinereus have not previously been documented. This seems unusual considering the importance of Elymus in the western United States and more specifically, in the Basin and Range Province (Lesperance et al. 1978). Whereas Elymus is susceptible to damage from defoliation (Krall et al. 1971; Perry & Chapman 1975), we found that removal of residual materials by burning had a positive effect on subsequent growth, when accomplished during dormancy. This may have important implications with respect to grazing management of this species. The presence of residual materials, coarse culms and inflorescences frequently results in poor utilization of Elymus by livestock (Lesperance et al. 1978). Dormant season burns, therefore, may be a useful means of increasing palatability and availability of Elymus as a forage species (Daubenmire 1968); and additionally, to increase control of the grazing animal and its subsequent effect on plants by improving herbage utilization (better animal distribution, reduced selectivity, etc.; Vallentine 1971).

Increased herbage production, shoot density and reproductive allocation are responses that likely will vary given similar burns in different years or different seasons of the year (Wright & Bailey

1982). Care should be taken in applying these results to other sites; especially to more xeric environments. Whereas increased flowering activity has been observed in many grass species after a fire (Burton 1944; Curtis & Partch 1950; Kucera & Ehrenreich 1962), herbage production, shoot response and the length (years) of observed response are highly variable (Wright & Bailey 1982).

Responses of the many species composing the CHNA/ELCI community appeared consistent in the field, i.e. increased abundance for one or two years. However, such differences are frequently difficult to demonstrate statistically. Fire's effect on annuals at these sites perhaps best typifies this situation. Individual plants of Bromus tectorum as well as the annual forbs were large and vigorous the year after the burns (personal observation), and ASC appeared to have been much less on non-use sites. Despite this, statistically significant differences were not observed. However, of those annuals identified at these sites (Appendix F), our general observation was that all benefited by the burns for one or both years thereafter. These observations are consistent with the findings of numerous researchers (Table I.1).

Similarly, Chrysothamnus spp. have been observed to increase after burns as a result of shrub survival/regrowth and seedling establishment (Young & Evans 1974). This has led to a projection of a minimum fire frequency of 25 to 35 years in the Artemisia zone of this region (Wright et al. 1979). We observed no increase in plant density

via seedling establishment, despite vigorous flowering in both years after the burns.

Previous reports of increased dominance by Chrysothamnus in postburn communities stem from observations in degraded communities, where perennial grasses were poorly represented. In contrast, perennial grasses dominated our study area (Appendices C and F). The potential differences in community development following disturbances in "good" vs. "poor" condition rangelands (Vallentine 1971) lead us to speculate that prior estimates of fire frequency in these habitats may be inaccurate if estimates are based upon responses observed in plant communities with little compositional similarity to those present prior to the arrival of white men in these areas.

#### ELTR Community

Many of the comments given above apply equally well to the observed fire effects in the ELTR vegetation type. Herbage production response, stem density and reproductive allocation was similar to responses observed in other North American mesic grasslands (Ehrenreich 1959; Kucera & Ehrenreich 1962; Hulbert 1969; Hadley 1970). Dormant season burns frequently increase herbage production, and frequently for only one to three years after a burn; seedhead production is often increased; and, increases in numbers of plants or shoots of rhizomatous species often occur (see reviews by Daubenmire 1968; Wright & Bailey 1982). In contrast, however, Old (1969) observed reductions in production, plant vigor, and plant numbers following a

burn in mesic grassland vegetation. This points out the importance of replicating burns over a number of years before drawing general conclusions about observed results.

As in the CHNA/ELCI community, no species was observed to have been harmed by burning. However, statistical support for this statement can be drawn for very few of the species observed.

#### SAVE/DISP Community

The limitations of results from this aspect of our studies restrict the conclusions that may be drawn. Response of Sarcobatus was similar to shrubs in the CHNA/ELCI habitat. Again, regrowth from basal sprouts was vigorous and indicated no lasting effect on individual shrubs.

Distichlis response was equally dramatic; this especially in light of the magnitude of production increase in a species well-adapted to stressful environments (Kemp & Cunningham 1981). Other references to Distichlis response to burning are confounded by the effects of flooding (Smith & Kadlec 1985). Our own studies, being limited to a single burn and vegetation responses observed for a single year, permit us to say only that the effects reported here are consistent with the responses we measured for a variety of associated emergent and terrestrial species. Therefore, we project the effect of fire on this species is similar to the others for which we have reported.

### Implications

Results of these studies point out the potential for manipulating these habitats through the use of controlled burns. Dormant season burning "improved" several aspects of plant growth. These findings demonstrate the ability to alter community structure and function without inducing significant changes in species composition or dominance patterns. These changes appear to be short-lived. This is brought out in the number of species for which "increases" were of a single year's duration. Additionally, replacement of preburn residual materials suggests this biomass component may return to pretreatment levels in no more than three to four years. Litter accumulations have been shown to be a major influence on community production, as well as composition and abundance of individual species in mesic grasslands (Wright & Bailey 1982). Furthermore, fire effects in these communities show remarkable similarities to results of these studies.

The question of long-term effects of periodic fires, occurring at shorter intervals, cannot be answered by our findings. But it is interesting to consider, and ultimately important to determine, how repeated burns at intervals of five to ten years would influence these same communities. For instance: if Chrysothamnus spp. and/or Sarcobatus require five or more years to fully recover from a burn; and, if perennial grasses are immediately favored by the same burn; then, is there a cumulative, positive effect of repeated fires on the herbaceous component, and, will this ultimately be manifested in directional changes in species composition/abundance? Our results

only suggest a positive response to this question. This is a major question with respect to fires historical role in shrub-grass habitats, and especially in those with shrub species capable of basal, root or epicormic sprouting after fire; e.g. Artemisia shrub-steppe, Arizona chaparral, southwest semidesert shrub-grasslands. This is also a question that likely, can only be addressed by long-term research/management efforts.

SECTION V

EFFECT OF DORMANT SEASON  
PRESCRIBED BURNING ON CIRSIIUM ARVENSE

Richard P. Young

Richard F. Miller

EFFECT OF DORMANT SEASON PRESCRIBED BURNING  
ON CIRSIIUM ARVENSE

SUMMARY

(1) Effects of dormant season burning (winter and early-spring) and non-use were evaluated in populations of Cirsium arvense and the associated vegetation of a mesic herbaceous community at Malheur National Wildlife Refuge, southeast Oregon. The objective of this study was to investigate the potential use of fire in controlling Cirsium invasion in this community. Cirsium response to fire was determined with respect to phenological development, height growth, shoot density, floral reproductive success, and herbage production. Herbage production of the associated vegetation was also determined.

(2) Winter and early-spring burns differed little in any of the parameters evaluated.

(3) Only minor differences in phenological development were observed with burning or non-use. Early in the growing season, height growth was slow and comparable among all treatments. However, during the period of rapid internode elongation height of non-use shoots increased more rapidly than shoots in burned areas. Maximum shoot height was achieved by non-use plants emerging early in the growing period.

(4) Herbage production of Cirsium was unaffected by burning. Associated vegetation, however, was most productive on burn areas. Non-use produced heavier and greater numbers of Cirsium shoots.



(5) Burning resulted in fewer total and fewer functional flower heads on reproductive shoots.

(6) Changes in population structure and reproductive success, as well as the differential increase in production of associated species, suggests burning increases intra- and interspecific interference between Cirsium and other members of this community. Dormant season burning reduced relative abundance of Cirsium and may be useful as a means of halting its invasion or spread by maintaining a productive stand of native vegetation.

## INTRODUCTION

Cirsium arvense (L.) Scop. (Canada thistle) is an aggressive perennial weed that reproduces from seed and rhizomes. Introduced from Europe, it is now widespread throughout much of the United States and Canada. Cirsium is well adapted to a variety of habitats and is responsible for substantial reductions in crop yields and both the quality and quantity of forages (Hodgson 1968; Hemmings & Toorn 1970 -- cited in Toorn & Mook 1982). It is a serious problem on some rangelands, especially highly productive, mesic sites where frequency (Stachon & Zimdahl 1980) and production (Reece & Wilson 1983) of native species can be severely reduced.

Extensive infestations of Cirsium occur along roadsides, canals, and sloughs, and extend into mesic herbaceous communities at Malheur National Wildlife Refuge (NWR) in southeastern Oregon. Refuge management practices designed to enhance waterfowl production, such as delayed haying dates and non-use of fields, contradict recommended cultural control methods (Hodgson 1958). Use of herbicides is restricted due to negative impacts on broadleaf dicots (Blaisdell & Mueggler 1956; Thilenius et al. 1974), and thus reduced quality of wildlife habitat (House et al. 1967).

Hemmings & Toorn (1970 -- cited in Toorn & Mook 1982) state Cirsium invasion of Phragmites australis (Cav.) Trin. ex Steudel (common reed) stands in the Netherlands can be prevented by burning old stems and litter in the winter. The purpose of this study was to evaluate Cirsium response to controlled burning within infested mesic

herbaceous communities at Malheur NWR. More specifically, we were interested in the effects of dormant season prescribed fire on certain structural and reproductive aspects of Cirsium populations, and on associated vegetation.

#### STUDY AREA

Malheur NWR is located approximately 40 km south of Burns in southeastern Oregon. The refuge lies within Harney Basin, a closed watershed occupying the northwestern-most portion of the Basin and Range Province (Baldwin 1964). Dominant features include Malheur and Harney Lakes in the northern portions of the refuge and the lower Blitzen River Valley, a major watershed of the basin, to the south. With the exception of open waters, the bulk of the refuge consists of flat to gently rolling landscapes (elevation: 1250-1280 m) derived from lake sediments and alluvium (Walker & Swanson 1968). Typical of high elevation basins of this region, the climate is semiarid with warm, dry summers and moist, moderate-cold winters (Gomm 1979). Annual precipitation averages 25-30 cm over the refuge.

The field chosen for study is located in the southern portion of the Blitzen River Valley. Lowlands in this area comprise a mosaic of wetland and terrestrial plant communities. Extensive wet meadows (syn.: flood meadows) and mesic herbaceous uplands occur in this part of the valley. These fields have been managed for native hay production and/or grazing since before the turn of the century (Griffiths 1902). They appear to be susceptible to invasion by

Cirsium, as many currently support large populations and some, dense stands (personal observation).

The study area is a mesic herbaceous community, a major upland vegetation type on the refuge (C. Rouse; unpublished data and maps on file at Malheur NWR, US Fish & Wildlife Service, 1961). Elymus triticoides Buckley (creeping wildrye) dominates the plant community. Other important species include Carex douglasii F. Boott (Douglas sedge), Juncus balticus var. montanus Engelm. (baltic rush), and Poa nevadensis Vasey ex Scrib. (Nevada bluegrass). A diverse mixture of other graminoids and forbs is associated with these species.

These are potentially, highly productive plant communities, capable of yielding in excess of  $350 \text{ g m}^{-2}$  of herbage annually (Young, unpublished data). This field had been managed for non-use for five years prior to initiation of the study. Typical of non-use fields, a large portion of the aboveground standing crop biomass consisted of litter and standing dead plant materials. Herbage production in 1980, the last growing season of the non-use period, was  $142 \text{ g m}^{-2}$ ; however, residual organic matter accounted for another  $632 \text{ g m}^{-2}$  of biomass. Cirsium populations, ranging from scattered shoots to dense, monotypic patches, occur throughout the study area.

## METHODS

Treatments consisted of winter and spring prescribed burns and non-use. Within each of the three treatment units five sites or replicates were selected on the basis of visual inspection. The

intent was to choose areas representing the range of moderate to heavy Cirsium infestations typical of wet meadows and mesic upland vegetation on the refuge. Both density and biomass were used as selection criteria, and in all cases Cirsium was the dominant or codominant species of the site. Each site was sufficiently large to permit delineation of a plot 10 x 10 m, judged homogenous with respect to composition of Cirsium and the associated species.

Both the winter (10 December 1980) and spring (7 April 1981) burns were conducted while vegetation was dormant. Weather conditions during both burns were cool with moderate to high relative humidity (Table V.1). Large quantities of continuous herbaceous fuels and steady, light winds (Table V.1), however, produced uniform, steadily advancing fires. Headfires were used as the ignition method (Martin & Dell 1978). Based upon rate of spread observations of the flaming front, and following the procedure of Alexander (1982), fire intensities of the winter and spring burns were  $1004 \text{ kW m}^{-1}$  and  $4465 \text{ kW m}^{-1}$ , or low and moderate intensity fires, respectively (McArthur & Cheney 1966). Despite differences in fire intensity, both burns consumed virtually all aboveground plant residues, leaving only white ash where formerly, dense patches of Cirsium had existed.

The shoots of a rhizomatous species constitute a population of vegetative structural units, or ramets. Although it is usually impossible to relate individual ramets to their respective parent plants, the genets, measurements on individuals and cohorts of shoots provide meaningful information relating species response to

Table V.1. Fuel characteristics and weather conditions at the time of winter (10 Dec. 1980) and spring (7 April 1981) controlled burns in a Cirsium infested mesic herbaceous community at Malheur NWR, southeast Oregon.

|                                   | Winter burn | Spring burn |
|-----------------------------------|-------------|-------------|
| Fuels                             |             |             |
| Loading ( $\text{g m}^{-2}$ )     | 644(145)*   | 473(160)    |
| Moisture content (%)              | 17          | 6           |
| Height (cm)                       | 25          | 20          |
| Weather                           |             |             |
| Temperature (C)                   | 2           | 6           |
| Relative humidity (%)             | 53          | 38          |
| Wind speed ( $\text{km h}^{-1}$ ) | 4-7         | 8-12        |

\* Mean  $\pm$  (standard deviation)

environment or treatment effects (Harper 1977). In this study Cirsium shoots are treated as a population of vegetative units which are used to relate treatment effects to the population of Cirsium plants as a whole.

Shoot height and phenological development were measured at 10-14 day intervals during the 1981 growing season. Detailed observations were obtained from marked stems (N=20 per treatment), and more generally, from the population of shoots at each site. Shoots selected for repeated observations were chosen on 28 April, and represented the range of shoot sizes present at that time. Shoot emergence continued throughout the growing season and, therefore, the marked shoots constitute an early cohort within the population studied. Phases of vegetative and floral reproductive conditions (Fig. V.1) were assigned to each shoot, permitting an estimate of phenological status for the population of Cirsium plants.

Several structural and developmental parameters were sampled within the Cirsium populations during the period of peak flowering activity. Shoot height measurements indicated peak flowering coincided with maximum herbage standing crop. Samples were obtained from four randomly located quadrats, 1.0 x 1.0 m, per replicate. Each Cirsium shoot in a quadrat was classified and tallied as either vegetative or florally reproductive, based upon visual observation of floral activity. Green buds, flowering heads, and heads with maturing seed were treated collectively as functional heads, and were counted on all reproductive shoots. Non-functional or aborted heads, which

were brown and commonly of reduced size, were tallied separately. These counts provide information on total, vegetative, and reproductive shoot densities; and for reproductive shoots, numbers of total, functional, and aborted heads.

Herbage within each quadrat was harvested, separating Cirsium from a composite sample of all other species. Litter was also collected at the non-use sites. Samples were subsequently oven-dried at 60 C for 48 hr and weighed.

Seasonal growth curves were prepared from data on shoot heights. Similarly, phenological observations were summarized graphically to illustrate Cirsium development through the growing season. Treatment effects on shoot height at three times during the growing season, representing phases of low and high growth rates (slow and rapid internode elongation and vegetative maturity), shoot density, herbage production and flowering success were evaluated using analysis of variance for a one-way classification. Where treatment effects were significant ( $P < .05$ ), least significant difference (lsd) tests were conducted for all possible pairwise comparisons (Steel & Torrie 1980).

## RESULTS

### Shoot Height Growth and Phenological Development

Newly emerged Cirsium rosettes were first noted on the winter burn area on 8 April, two days before the spring burn was conducted, and shoots were present in abundance on all treatment areas by 18



April (Fig. V.1). With the exception of this initial difference in emergence, winter and spring burns were not discernably different with respect to height or development of Cirsium over the remainder of the season; and therefore, they are considered as a single burn treatment in the discussion of these topics.

Height growth in the rosette stage was slow in all treatments and achieved through the addition of new leaves with little internode elongation. Internode elongation became apparent after rosettes reached the 4-6 leaf stage, which occurred among the larger shoots in late April to early May in all treatments. A period of slow internode elongation was observed until early June, during which the rate of height growth and addition of new leaves were similar among treatments. Although non-use shoots were generally taller at this time (17 May: non-use=12.7 cm; winter burn=9.8 cm; spring burn=9.4 cm), height differences were not significant ( $P>.05$ ). Growth rates increased dramatically in early to mid-June, continuing until mid-July. This was apparently a result of rapid internode elongation. Non-use shoots grew most rapidly during this period, displaying markedly greater stature ( $P<.05$ ) ( $\bar{x}$ =30.2 cm), than winter and spring burned shoots ( $\bar{x}$ =23.0 and  $\bar{x}$ =27.6 cm, respectively) on 21 June.

Flower buds were evident among non-use shoots by 3 July, and were abundant by 14 July. Some flowering occurred during the next few weeks, although many buds were aborted. Bud formation in burn treatments was delayed an additional 1-2 weeks. Even after abundant buds were present, most heads were aborted, and few functional

Fig. V.1. Height growth curve and phenological development of Cirsium shoots during the 1981 growing season; boundaries of phenological stages are approximate as the classes are nonexclusive and have been generalized to include all treatments; heights are derived from shoots that emerged early in the growing season. Differences between non-use and burn responses are noted for three dates with  $1sd^{5/17}$  (4.4),  $1sd^{6/21}$  (5.7) and  $1sd^{8/8}$  (6.2); O---O, non-use, N=20; \*.....\*, burn, N=40.

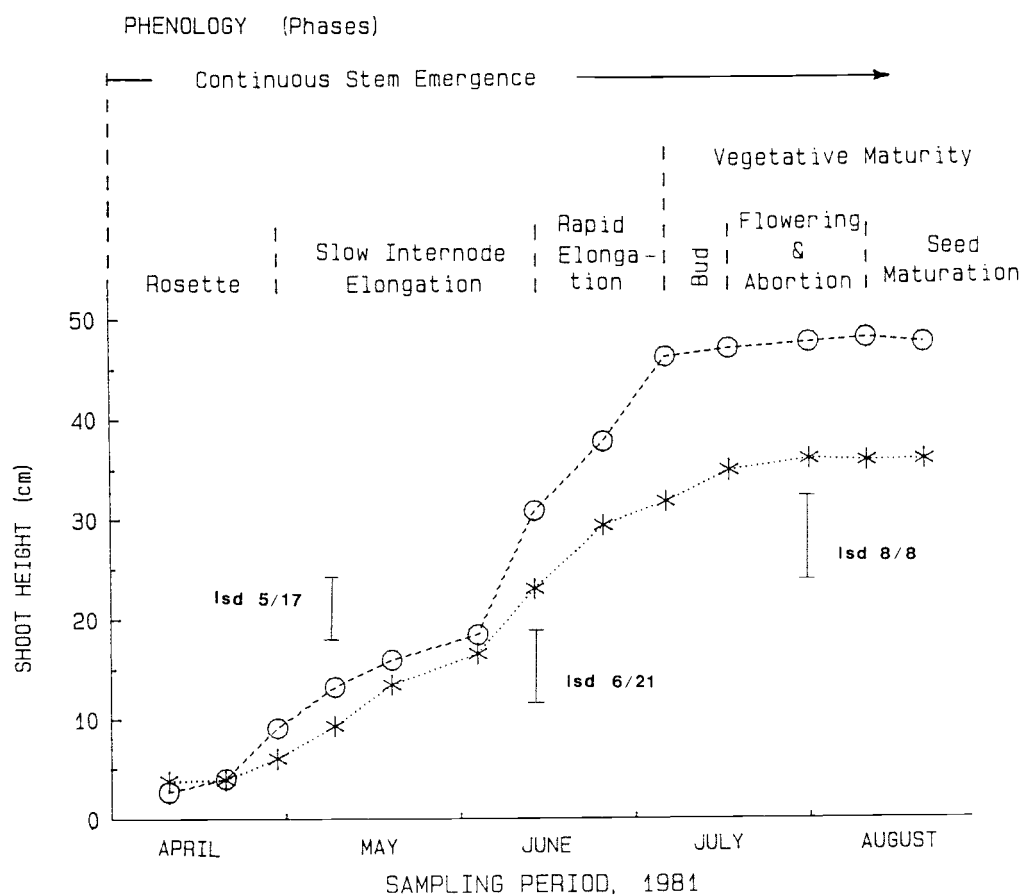


Fig. V.1

flowers were produced on the burns. Flower buds and heads were inspected for insect damage; however, no evidence was found linking insect activity to the high percentage of aborted heads. Peak floral activity occurred from early to mid-August, and flowering had essentially terminated on all sites by late August.

Height growth slowed appreciably after initiation of floral activity. By the time of peak flowering activity, the larger shoots of all treatments had achieved maximum height. A comparison of marked shoot heights for 8 August indicated no difference in the sizes of winter ( $\bar{x}=35.5$  cm) and spring ( $\bar{x}=34.7$  cm) burned plants ( $P>.05$ ); however, non-use shoots were much taller ( $\bar{x}=46.5$  cm) than in either burn treatment ( $P<.05$ ). These differences reflect the growth pattern of the early cohort of marked shoots, all of which developed through the flowering and/or bud abortion phenological stages. However, new shoots continued to emerge throughout the growing season in all treatments, resulting in a continuum of plant heights within each population of shoots. Late-emerging shoots largely remained vegetative, producing no functional flowers or viable seed.

#### Density and Biomass Production Relationships

Timing of dormant season fires in mesic herbaceous vegetation did not affect subsequent shoot density (total, vegetative or reproductive shoots)(Table V.2) or biomass production of Cirsium arvense, production of the associated vegetation, nor, therefore, total community aerial primary production (Table V.3), through the first postburn growing

Table V.2. Total Cirsium shoot density ( $m^{-2}$ ), vegetative and reproductive components of density, and the ratio, vegetative: reproductive shoots; seasonally burned and non-use treatments in a mesic herbaceous community, Malheur NWR. Sampling coincided with, estimated, peak live standing crop, in mid-August. Within columns, means followed by the same letter are not significantly different ( $P>.05$ )

| Treatment   | Total | Vegetative | Reproductive | Ratio |
|-------------|-------|------------|--------------|-------|
| Non-use     | 116a  | 55a        | 61a          | 0.9   |
| Winter burn | 261b  | 206b       | 55a          | 3.7   |
| Spring burn | 245b  | 181b       | 64a          | 2.8   |

Table V.3. Aboveground standing crop ( $\text{g m}^{-2}$ ; live and recent dead) of Cirsium arvense and all other species combined; for seasonally burned and non-use treatments in a mesic herbaceous community, Malheur NWR. Sampling coincided with, estimated, peak live standing crop (see text for discussion of this point), in mid-August. Within columns, means followed by the same letter are not significantly different ( $P > .05$ ).

| Treatment   | <u>Cirsium</u> | Other vegetation | Total |
|-------------|----------------|------------------|-------|
| Non-use     | 253a           | 53a              | 306b  |
| Winter burn | 286a           | 116b             | 402b  |
| Spring burn | 274a           | 114b             | 388b  |

season. (These results suggest treatment affects on the Cirsium dominated vegetation of concern here, may justifiably and productively be viewed in the context of the role of fire, applied to dormant vegetation, in contrast to the influence of non-use in these communities. This approach is followed in the remainder of this paper, with reference to seasonal influences, hopefully, wherever it is instructive to do so.)

Fire stimulated Cirsium shoot production with respect to the number of vegetative structures. Total shoot density within burned areas increased to levels greater than two times the density of shoots on non-use plots (Table V.2). Despite this, densities of reproductive shoots were similar and treatment differences in total shoot density were a function of the greater abundance of vegetative shoots on sites modified by fire. Non-use Cirsium populations were comprised of nearly equal numbers of vegetative and reproductive shoots. On burned sites vegetative outnumbered reproductive shoots by a factor of almost 3-4 times (Table V.2).

Fire increased total herbage production (measured here as the aboveground standing crop of live and recent dead tissues) in the first postburn growing season (Table V.3). Peak standing crop of Cirsium after fire was similar to standing crops in unburned areas, and therefore, variation in total herbage production was attributed to increased production by the associated vegetation in burned areas.

Differences in production may be expressed in terms of mean plant weights. Mean Cirsium shoot biomass was determined from

herbage production and total shoot densities for each treatment. Calculated values indicate that, on the average, non-use shoots were nearly twice as large as shoots of either burn treatment: non-use=2.20 g; winter burn=1.10 g; and, spring burn=1.12 g.

#### Flowering Success

The total number of heads produced (among plants achieving floral reproductive status) and the number of functional heads that developed were similar among the winter and spring burns (Table V.4). Relative to non-use, however, burning produced significantly fewer total and functional heads per reproductive shoot. There were no differences among treatments with respect to numbers of aborted heads. On burn areas >10% of all heads produced remained functional, compared to >45% on the non-use area.

#### DISCUSSION

Cirsium response to burning consisted of both positive and negative aspects. Although shoot density increased with burning both average shoot weight and height of the early cohort of shoots was greatest with non-use. Burning, therefore, altered the population structure from fewer, larger ramets, to a higher density of smaller shoots. Total herbage production of Cirsium was unaffected by either this structural change or any other effects on the plant community attributable to burning (see reviews by Daubenmire 1968; Wright & Bailey 1982). Concurrent studies at Malheur NWR repeatedly showed



Table V.4. Total number of flower heads (= inflorescences) per Cirsium shoot, functional and aborted components of the population of seedheads, and the ratio, functional: aborted heads, for seasonally burned and non-use treatments in a mesic herbaceous community, Malheur NWR. Sampling coincided with the period of peak floral activity, early to mid-August. Within columns, means followed by the same letter are not significantly different ( $P > .05$ )

| Treatment   | Total | Functional | Aborted | Ratio |
|-------------|-------|------------|---------|-------|
| Non-use     | 36.2a | 16.3a      | 19.9a   | .819  |
| Winter burn | 18.3b | 1.2b       | 17.1a   | .070  |
| Spring burn | 17.2b | 0.9b       | 16.3a   | .055  |

rhizomatous monocots, respond to dormant season burning with increased shoot density, total herbage production, and frequently mean shoot weight. Vegetation associated with Cirsium displayed a similar trend towards increased herbage production after burning. Cirsium was the only major species, therefore, which did not increase biomass production, following dormant season prescribed burning.

The differential response of Cirsium and of the associated species (as a group), with regards to herbage production, indicate Cirsium was subject to increased levels of interspecific interference. Furthermore, the greater density of smaller shoots suggest intraspecific interference increased among the population of ramets. Both these notions are supported by Cirsium floral reproductive response. Reproductive effort, as measured by number of flowering plants, flowers per plant, and/or seed or fruit yield have been shown to be related to plant density, and therefore competition, in single-species stands (Clements et al. 1929; Hodgson & Blackman 1957) and multi-species stands (Palmbad 1968) of annuals, biennials (Oxley -- cited in Harper 1977), rhizomatous perennials (Ogden 1970), and trees (Verheij 1968). Typically, beyond some population density level, the biomass produced becomes limited by resource availability and reproductive effort or yield declines (Harper 1977). Competition for limited resources may be either or both intraspecific or interspecific. In this study burning increased production of associated vegetation, increased Cirsium shoot density and resulted in fewer flower heads on Cirsium plants. Therefore, floral reproductive effort and success

decreased in response to burning. Prescribed burning during the dormant season did not decrease Cirsium biomass production and, therefore, may not be effective in eliminating Cirsium where it has become established. Fire did, however, reduce the relative abundance of Cirsium within these communities. Furthermore, the potential of spreading by seed produced within an established population was reduced due to the negative impact of fire on reproductive success. The response of Cirsium populations and the associated vegetation to fire suggest Cirsium invasion may be slowed or contained by this treatment. By increasing productivity of native vegetation, it may also be possible to prevent Cirsium invasion into new habitats. This possibility is supported by the findings of Hemminga & Toorn (1970) in which winter burning maintains the productivity of Phragmites australis communities and prevents Cirsium invasion.

Although fire did not control Cirsium in the manner of previously developed cultural and chemical treatments, it may be a useful management tool where these methods can not be implemented. Based upon these results, the use of repeated prescribed fires warrants further study as a potential Cirsium control measure not formerly employed by land managers.

## SECTION VI

### CONCLUSIONS AND MANAGEMENT IMPLICATIONS

## CONCLUSIONS AND MANAGEMENT IMPLICATIONS

This research was structured to provide the information base needed to develop fire management plans suitable for wetland and upland habitats of Malheur NWR. It was presumed from the outset that fire, when and where properly applied, is capable of accomplishing a number of wetland ecosystem management objectives. This has been documented in reports of prescribed burning from a number of wetland studies (see reviews in Section I), with most uses of fire for the purpose of wildlife management. The information required for fire planning in inland wetlands of the western United States has previously been unavailable. This study addressed those needs with the intention of improving our understanding of fire behavior and fire effects, specific to these habitats. Certainly we have much still to learn. However, results of these investigations are sufficient to produce fire management plans based upon ecological principles (Kessell & Fischer 1981), knowing these undoubtedly will be modified as experience and additional studies improve our knowledge of these ecosystems.

Results of our fire effects studies illustrate some of the influences of burning on wetland plant communities. Specific responses we observed (eg. increased aerial standing crop, shoot density, etc.) should not be expected for every controlled burn. Fire effects are often the result of the specific context in which burning occurred, and possibly more important yet, postburn influences such as water management, grazing and haying. Although fire is a potentially

powerful force in wetland ecosystems, many reports illustrate the overwhelming influence of water management on wetlands (Kadlec 1962; Harris & Marshall 1963; Meeks 1969; Tesky & Hinkley 1977; Bishop et al. 1979). Just as fire research in terrestrial systems has produced seemingly contradictory results, observations from wetland burns are not wholly consistent; and reported differences probably best serve to illustrate what we do not know with respect to the mechanisms at work. A "high degree" of similarity however, exists among observations of fire effects in wetland vegetation; and results of this study support many aspects of Vogl's (1980) summary on this topic.

The community dynamics of wetland vegetation and the influence of fire in affecting those dynamics are of special importance, both from a purely ecological perspective and in terms of applying these principles to management situations. The long-term stability of wetland systems seems to contradict the obvious characteristics of high productivity and net accumulation of organic residues which have lead to the emphasis of hydrarch successional principles. Fire, as a process in these systems helps explain this dilemma. Not only does fire act as a "rejuvenating" force in wetlands, the adaptations of rhizomatous emergent macrophytes (especially, meristems protected from heat damage and continuous occupation/dominance after fire) are examples of fire-dependent communities hypothesized by Mutch (1970).

Results of dormant season burns in upland communities suggest these habitats respond in a manner similar to adjacent wetlands, than

the more xeric terrestrial communities of the Artemisia and Atriplex zones. This is not unexpected, based upon the floral composition of the upland communities: many of the dominant graminoids are components of floodmeadow and emergent communities at Malheur; and, Elymus cinereus and many of the subordinate graminoids common to the regional semidesert flora, are most abundant in mesic ecosystems of these high-desert rangelands. Thus it seems logical that prescribed burning during the dormant seasons should produce vegetation responses similar to those of emergent wetlands and mesic grassland communities.

Vegetation response to fire on the xeric terrestrial sites is well documented. Documented responses are much more variable, indicated by the number of species damaged by fire and the long recovery periods. Such results should not be ruled out as potential outcomes of burns conducted in the mesic uplands of Malheur, when these occur during periods of plant growth and/or warmer, drier weather.

A limitation of this, as in many studies, was the inability to evaluate long-term dynamics; and especially in the context of repeated occurrence of burning. Future efforts in fire research in wetlands should approach this major question. Fire behavior research should be directed towards extending the limits of prescriptions and recommendations proposed in this report, especially with respect to: (1) season of burning in wetlands and adjacent uplands; and, (2) parameters that most strongly influence fire behavior in these

habitats, including the effects of wind speed, temperature and fuel surface moisture.



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## APPENDICES

Appendix A. Pretreatment characteristics of five wetland plant communities at Malheur NWR, August 1980. Data were used to test the assumption of homogeneity, within a community type, among sites selected for burning and non-use treatments. Sampling procedures: (1) aboveground standing crop -- harvested from 10 quadrats, 0.5 x 0.5 m, per treatment (2 per replicate x 5 replicates), and separated into live and dead components; (2) shoot density -- all live shoots occurring in the 0.25 m<sup>2</sup> quadrats were counted prior to harvesting biomass samples. Analyses of variance indicated no differences in standing crop or density attributes ( $P > .05$ ) among treatment study sites within any of the five communities.

| Community                                   | Treatment   | Live        |            |                               |
|---|-------------|-------------|------------|-------------------------------|
| <u>Scirpus acutus</u>                       | Non-use     | 776 (172) * | 1048 (247) |                               |
| Aboveground Biomass<br>(g m <sup>-2</sup> ) |             |             |            |                               |
| Community                                   | Treatment   | Live        | Dead       | Density<br>(m <sup>-2</sup> ) |
| <u>Scirpus acutus</u>                       | Non-use     | 776 (172) * | 1048 (247) | 86 (33)                       |
|   | Winter burn | 811 (205)   | 956 (211)  | 74 (21)                       |
|   | Spring burn | 783 (191)   | 1087 (235) | 90 (35)                       |
| <u>Sparganium eurycarpum</u>                | Non-use     | 350 (103)   | 425 (99)   | 64 (12)                       |
|   | Winter burn | 372 (127)   | 514 (158)  | 73 (20)                       |
|   | Spring burn | 406 (142)   | 477 (142)  | 61 (13)                       |
| <u>Carex atherodes</u>                      | Non-use     | 798 (154)   | 617 (124)  | 282 (57)                      |
|   | Winter burn | 788 (170)   | 725 (132)  | 236 (62)                      |
|   | Spring burn | 830 (165)   | 643 (98)   | 239 (52)                      |
| <u>Juncus balticus</u>                      | Non-use     | 382 (67)    | 708 (124)  | 986 (241)                     |
|   | Fall burn   | 419 (81)    | 678 (95)   | 1021 (267)                    |
| <u>Eleocharis palustris</u>                 | Non-use     | 375 (51)    | 218 (57)   | 1792 (422)                    |
|   | Fall burn   | 406 (63)    | 199 (46)   | 1870 (465)                    |

\* Mean  $\pm$  (standard deviation)

Appendix B. Aboveground standing crop of Elymus triticoides upland community study sites, 17-18 August 1980 (= pretreatment). Data were used to test the assumption of homogeneity among potential sites selected for burning and non-use treatments. Sampling procedures: collection of total aboveground plant material in 10 quadrats, 0.5 x 0.5 m, per treatment (2 per replicate x 5 replicates); all treatments combined, N=30. Analysis of variance indicated no differences among treatment study areas ( $P > .05$ ) for any standing crop component.

| Component            | Standing Crop<br>(g m <sup>-2</sup> ) |             |             | All sites<br>combined |
|----------------------|---------------------------------------|-------------|-------------|-----------------------|
|                      | Non-use                               | Winter burn | Spring burn |                       |
| Perennial graminoids | 84 (32)*                              | 77 (28)     | 95 (33)     | 85 (30)               |
| Perennial forbs      | 47 (20)                               | 64 (22)     | 59 (31)     | 57 (24)               |
| Annuals              | 0 (0)                                 | 2 (5)       | 0 (0)       | <1 (3)                |
| Total herbage        | 131 (39)                              | 141 (49)    | 154 (52)    | 142 (46)              |
| Residual materials   | 640 (175)                             | 553 (141)   | 702 (193)   | 632 (170)             |

\* Mean  $\pm$  (standard deviation)

Appendix C. Aboveground standing crop of herbaceous vegetation at the Chrysanthemum nauseosus/Elymus cinereus upland community study sites, 22-24 July 1980 (= pretreatment). Data were used to test the assumption of homogeneity among potential sites selected for burning and non-use treatments. Sampling procedures: aboveground plant materials, exclusive of live shrubs or woody debris, harvested from 20 quadrats, 0.5 x 0.5 m, per treatment (4 per replicate x 5 replicates); all treatments combined, N=60. Analyses of variance indicated no differences among treatment study sites ( $P>.05$ ) for any standing crop component.

| Component            | Standing Crop<br>(g m <sup>-2</sup> ) |             |             | All sites<br>combined |
|----------------------|---------------------------------------|-------------|-------------|-----------------------|
|                      | Non-use                               | Winter burn | Spring burn |                       |
| Perennial graminoids | 186 (119)*                            | 132 (83)    | 144 (97)    | 154 (99)              |
| Annual grasses       | 3 (6)                                 | 14 (23)     | 5 (8)       | 7 (12)                |
| Perennial forbs      | 1 (3)                                 | 8 (17)      | 4 (9)       | 4 (9)                 |
| Annual forbs         | 4 (6)                                 | 8 (15)      | 7 (12)      | 6 (10)                |
| Total herbage        | 194 (120)                             | 162 (88)    | 160 (101)   | 172 (104)             |
| Residual materials   | 427 (241)                             | 313 (172)   | 338 (161)   | 359 (191)             |

\* Mean  $\pm$  (standard deviation)



Appendix D. Aboveground standing crop of herbaceous vegetation at the Sarcobatus vermiculatus/Distichlis spicata upland community sites, 3 September 1980 (= pretreatment). Data were used to test the assumption of homogeneity among potential sites selected for burning and non-use treatments. Sampling procedures: collection of aboveground plant materials, exclusive of live shrubs or woody debris, in 16 quadrats, 0.5 x 0.5 m, per treatment (2 per replicate x 8 replicates); all sites combined, N=48. Analysis of variance indicated no differences among treatment study sites ( $P > .05$ ) for any standing crop component.

| Component            | Standing Crop<br>(g m <sup>-2</sup> ) |             | All sites<br>combined |
|----------------------|---------------------------------------|-------------|-----------------------|
|                      | Non-use                               | Spring burn |                       |
| Perennial graminoids | 345 (105)*                            | 427 (88)    | 386 (94)              |
| Other species        | 21 (29)                               | 13 (25)     | 17 (30)               |
| Total herbage        | 366 (121)                             | 440 (110)   | 403 (102)             |
| Residual materials   | 585 (273)                             | 738 (288)   | 661 (291)             |

\* Mean  $\pm$  (standard deviation)

Appendix E. Frequency of occurrence (%) of plant species in the Elymus triticoides upland community, August 1980 (=pretreatment). Measured as the proportional occurrence of plants rooted within quadrats 0.5 x 0.5 m; 10 quadrats per 10 m line transect, 4 transects per treatment. Species are classified by functional groups = group habit. Symbols; +: present in the experimental unit, but not occurring within sample quadrats; -: not found in the experimental unit, but occurring in other units of the community type.

#### Growth Habit

| Species  | Non-use | Winter<br>burn | Spring<br>burn | All Sites |
|--|---------|----------------|----------------|-----------|
| Perennial Grasses & Grasslikes                   | 100     | 100            | 100            | 100       |
| <u>Carex aquatilis</u>                           | -       | +              | +              | -         |
| <u>Carex douglasii</u>                           | 100     | 100            | 100            | 100       |
| <u>Distichlis spicata</u><br>var. <u>stricta</u> |         |                |                | 3         |
| <u>Elymus triticoides</u>                        | 100     | 100            | 100            | 100       |
| <u>Hordeum brachyantherum</u>                    | 1       | +              | 1              | 1         |
| <u>Hordeum jubatum</u>                           | +       | +              | 1              | 1         |
| <u>Juncus balticus</u><br>var. <u>montanus</u>   | 86      | 93             | 88             | 89        |
| <u>Muhlenbergia asperfolia</u>                   | 7       | 4              | 2              | 4         |
| <u>Poa nevadensis</u>                            | 2       | 2              | 6              | 3         |
| <u>Poa pratensis</u>                             | +       | -              | -              | +         |
| Perennial Forbs                                  | 49      | 53             | 56             | 53        |
| <u>Achillea millefolium</u>                      | 7       | 4              | 8              | 6         |
| <u>Agoseris glauca</u>                           | 12      | 10             | 17             | 13        |
| <u>Artemisia ludoviciana</u>                     | +       | +              | 2              | 1         |
| <u>Aster chilensis</u><br>ssp. <u>adscendens</u> | 22      | 31             | 28             | 27        |
| <u>Cirsium arvense</u>                           | 10      | 6              | 3              | 6         |
| <u>Cirsium vulgare</u>                           | +       | +              | +              | +         |
| <u>Crepis runcinata</u>                          | 7       | 5              | 3              | 5         |
| <u>Haplopappus lanceolatus</u>                   | 32      | 24             | 29             | 28        |

## Appendix E. -- Continued

## Growth Habit

| Species  | Non-use | Winter<br>burn | Spring<br>burn | All Sites |
|--|---------|----------------|----------------|-----------|
| Perennial Forbs (Continued)                                |         |                |                |           |
| <u>Penstemon rydbergii</u>                                 | 2       | 1              | 2              | 2         |
| <u>Potentilla anserina</u>                                 | +       | 2              | 1              | 1         |
| <u>Potentilla gracilis</u>                                 | 3       | 4              | 2              | 3         |
| <u>Sidalcea neomexicana</u>                                | +       | 1              | +              | 1         |
| <u>Sisyrinchium angustifolium</u>                          | +       | +              | 3              | 1         |
| <u>Solidago canadensis</u><br>var. <u>salebrosa</u>        | 2       | 1              | +              | 1         |
| <u>Solidago missouriensis</u><br>var. <u>missouriensis</u> | +       | +              | +              | +         |
| <u>Taraxacum officinale</u>                                | +       | 1              | +              | 1         |
| <u>Thermopsis montana</u><br>var. <u>ovata</u>             | +       | +              | +              | +         |
| <u>Tragopogon dubius</u>                                   | 4       | 4              | 1              | 3         |
| <u>Vicia americana</u>                                     | 2       | 2              | 1              | 2         |
| Annual Forbs   | 2       | 2              | 2              | 2         |
| <u>Chenopodium album</u>                                   | +       | +              | 1              | 1         |
| <u>Chenopodium</u> sp.                                     | +       | -              | -              | +         |
| <u>Descurainia pinnata</u>                                 | 1       | 2              | +              | 1         |
| <u>Lactuca pulchella</u>                                   | 2       | 1              | 1              | 1         |
| <u>Lepidium perfoliatum</u>                                | 1       | +              | +              | 1         |
| <u>Thlaspi arvense</u>                                     | +       | -              | +              | +         |

Appendix F. Plant species composition of the Chrysothamnus nauseosus/Elymus cinereus upland community, July 1980 (=pretreatment). Shrub canopy cover (%): line intercept; 4 transects, 10 m, per replicate x 5 replicates per treatment. Shrub density ( $\text{ha}^{-1}$ ): plants rooted within belts (= strip transects), 2 x 10 m; belts centered on, that is, 1 m width either side of the (above) 10 m line transects. Frequency of occurrence (%) of herbaceous species: proportional occurrence of plants rooted within quadrats 0.5 x 0.5 m; 10 quadrats per 10 m line transect. Cover and density values are  $\bar{x} \pm (\text{sd})$  of treatment study sites and the community type as a unit. Species are classified by functional groups = growth habit. Symbols; +: present in the experimental unit, but not occurring within sample quadrats; -: not found in the experimental unit, but occurring in other units of the community type; \*: see special note at the end of this appendix.

#### Growth Habit

| Species   | Measure of abundance | Non-use                | Winter burn            | Spring burn            | All Sites              |
|---|----------------------|------------------------|------------------------|------------------------|------------------------|
| <b>Shrubs</b>   |                      |                        |                        |                        |                        |
| <u>Artemisia tridentata</u><br>ssp. <u>tridentata</u>             |                      | +                      | -                      | -                      | +                      |
| <u>Chrysothamnus nauseosus</u><br>var. <u>albicaulis</u> *        | cover<br>density     | 5.8 (4.2)<br>327 (241) | 6.1 (4.9)<br>348 (218) | 5.6 (3.3)<br>265 (203) | 5.8 (4.3)<br>313 (227) |
| <u>Chrysothamnus viscidiflorus</u><br>var. <u>viscidiflorus</u> * | cover<br>density     | 2.6 (1.7)<br>156 (121) | 3.2 (2.8)<br>133 (94)  | 2.7 (2.0)<br>188 (150) | 2.8 (2.3)<br>159 (132) |
| <u>Sarcobatus vermiculatus</u> *                                  | cover<br>density     | 5.3 (3.5)<br>147 (115) | 4.9 (4.1)<br>119 (93)  | 4.4 (2.9)<br>128 (104) | 4.8 (3.7)<br>131 (106) |
| <b>Perennial Grasses &amp; Grasslikes</b>                         |                      |                        |                        |                        |                        |
| <u>Carex douglasii</u>  | frequency            | 93                     | 97                     | 94                     | 95                     |
| <u>Distichlis spicata</u> var. <u>stricta</u>                     |                      | 14                     | 18                     | 17                     | 16                     |
| <u>Elymus cinereus</u>  |                      | 4                      | 6                      | 3                      | 4                      |
|   |                      | 23                     | 28                     | 24                     | 25                     |

Appendix F. -- Continued

Growth Habit

| Species                                     | Measure of abundance | Non-use | Winter burn | Spring burn | All Sites |
|---|----------------------|---------|-------------|-------------|-----------|
| <u>Elymus triticoides</u>                   | frequency            | 61      | 68          | 70          | 66        |
| <u>Juncus balticus</u> var. <u>montanus</u> |                      | 5       | 8           | 7           | 7         |
| <u>Muhlenbergia asperifolia</u>             |                      | +       | +           | +           | +         |
| <u>Poa nevadensis</u>                       |                      | 5       | 7           | 8           | 7         |
| Annual Grasses                              | frequency            |         |             |             |           |
| <u>Bromus tectorum</u>                      |                      | 13      | 18          | 14          | 15        |
| Perennial Forbs                             | frequency            | 8       | 3           | 6           | 6         |
| <u>Arabis holboellii</u>                    |                      | 2       | 1           | 1           | 1         |
| <u>Artemisia ludoviciana</u>                |                      | 1       | 1           | 1           | 1         |
| <u>Astragalus filipes</u>                   |                      | +       | -           | +           | +         |
| <u>Astragalus purshii</u>                   |                      | 1       | -           | +           | 1         |
| <u>Cirsium arvense</u>                      |                      | 6       | 2           | 5           | 4         |
| <u>Cirsium vulgare</u>                      |                      | +       | +           | +           | +         |
| <u>Urtica dioica</u>                        |                      | +       | +           | +           | +         |
| Annual Forbs                                | frequency            | 16      | 18          | 13          | 16        |
| <u>Alyssum alyssoides</u>                   |                      | 2       | 1           | 1           | 1         |
| <u>Amsinkia tessellata</u>                  |                      | +       | +           | +           | +         |

Appendix F. -- Continued

Growth Habit

| Species   | Measure of abundance | Non-use | Winter burn | Spring burn | All Sites |
|---|----------------------|---------|-------------|-------------|-----------|
| <u>Chaenactis douglasii</u>                                 | frequency            | +       | 1           | +           | 1         |
| <u>Chenopodium</u> sp.                                      |                      | 1       | +           | +           | 1         |
| <u>Cordylanthus ramosus</u>                                 |                      | 1       | 2           | 1           | 1         |
| <u>Descurainia pinnata</u>                                  |                      | 8       | 12          | 10          | 10        |
| <u>Draba verna</u>  |                      | +       | 1           | 1           | 1         |
| <u>Gayophytum diffusum</u>                                  |                      | 6       | 2           | 3           | 4         |
| <u>Gilia sinuata</u>  |                      | -       | +           | -           | +         |
| <u>Lactuca pulchella</u>                                    |                      | 2       | 3           | 2           | 2         |
| <u>Lappula redowskii</u>                                    |                      | +       | +           | -           | +         |
| <u>Lepidium perfoliatum</u>                                 |                      | 6       | 8           | 3           | 6         |
| <u>Lesquerella occidentalis</u><br>var. <u>occidentalis</u> |                      | 1       | +           | 1           | 1         |
| <u>Phacelia linearis</u>                                    |                      | +       | 1           | +           | 1         |
| <u>Polygonum douglasii</u>                                  |                      | +       | 2           | 1           | 1         |
| <u>Salsola iberica</u>                                      |                      | +       | -           | -           | +         |
| <u>Sisymbrium altissimum</u>                                |                      | 2       | 2           | 1           | 2         |
| <u>Thelopodium laciniatum</u><br>var. <u>laciniatum</u>     |                      | +       | -           | +           | +         |