PESTICIDE SECTION

CANADIAN WILDLIFE SERVICE

MANUSCRIPT REPORTS

Ecological Considerations of Herbicide Use in Alberta

No. 26

J.H. Patterson

1973

CANADIAN WILDLIFE SERVICE

PESTICIDE SECTION
MANUSCRIPT REPORTS

Ecological Considerations of Herbicide Use in Alberta

J.H. Patterson
Research Scientist
Canadian Wildlife Service
Edmonton, Alberta
April, 1973

Contents

A. Environmental Contamination by Chlorophenoxy Herbicides

Each year about 27 million acres of cropland in the Canadian prairies are sprayed with phenoxy herbicides (Maybank and Yoshida, 1969). Of the approximately 20 million pounds of 2,4-D and MCPA applied to crops, considerable quantities can escape to non-target areas. The hazards of herbicide drift to agricultural crops are well recognized as reports of damage to box elder trees, rapeseed, sunflowers, sweet clover and shelterbelt trees are common. In south central Saskatchewan, for example, it is impossible to find a box elder tree in July that does not show symptoms of damage typical of that caused by 2,4-D (Hay and Grover, 1967).

During spraying operations the herbicides contribute micronized particles and/or volatiles to the atmosphere. Even after the herbicide has contacted the soil, plants or other objects, volatile chemicals (either the original herbicide or its degradation products) may reenter the atmosphere in vapour form (Foy and Bingham, 1970).

Maybank and Yoshida (1969) have demonstrated that under recommended spraying conditions in Saskatchewan, up to 30% of ground sprayed, 2,4-D is subject to drift.

Grover et al. (1972) experimented with 2,4-D drift and volatility at Suffield, Alberta. They showed that within the first half-hour after conventional ground spraying, up to 30% of the butyl ester was collected as a vapour drift, in addition to the drift. Therefore, it is by no means an exaggeration to say that over 50% of a 2,4-D ester application can go into the environment as a pollutant.

The only concern expressed by the agriculturalists is that herbicide movement may damage sensitive crops or ornamentals. In fact the Alberta Weed Control and Field Services Branch recommends spraying heavy doses of herbicides in stubble fields and along fence rows during the fall, as drift or volatility present no problems at this time (Anon., 1972).

It is often said that the biological significance of these very low concentrations in the air have not yet been established. The fact that there is documented damage to other crops, shelter-belts etc. indicates that there is conclusive biological significance. What is upsetting is the fact that there is no one looking for corresponding damage in the natural environment. Most of the noticeable plants, e.g. trees are relatively resistent to 2,4-D. Thus, when they are damaged, the dose must have been very high. Even when the trees are not damaged, how many shrubs and herbs have been affected without being noticed.

Because spray drift and atmospheric contamination are an annual fact of life in western Canada, perhaps one of the most serious and at the same time most subtle threats to wildlife resources from the agricultural use of herbicides is that of the unintended impact on non-target ecosystems.

In an attempt to evaluate the habitat damage resulting from herbicide contamination, not only toxicity but persistence have to be considered. For example, relatively similar degrees of phytotoxicity of decomposition can be strongly habitat specific. In terrestrial agricultural soils, degradation rates are generally quite rapid. Residues typically persist for only 1 to 4 weeks (Sheets and Harris 1965). In contrast, phenoxy herbicides can be very persistent in wetland ecosystems.

In laboratory experiments, Hemmett (1972) found that although 2,4-D acid was readily degraded by aquatic organisms, the isopropyl and butyl esters of 2,4-D were degraded at only half the rate, and phenoxy herbicides (3,4-D, 2,4,5-T, 2(2,4,5-TP), the isooctyl ester of 2,4-D, MCPA and 2-CPA) could not be degraded at all.

Not all aquatic environments are as favourable for herbicide degradation as those used by Hemmett. For example, DeMarco (1967) found that in cool deoxygenated waters, it took over 13 times as long to degrade 2,4-D than in warm aerobic waters. This fact is of particular importance in western Canada, as wetlands in this edaphically rich area are characterized by cool, deoxygenated bottom waters.

Of even greater environmental concern is the fact that herbicides can accumulate in aquatic sediments. For example, Smith and Isom (1971) reported on the application of up to 100 pounds of the butoxyethanol ester of 2,4-D per acre to reservoirs for water millfoil control. Eight hours after application, the highest amount found in the water was 64 parts per billion.

However, after 24 hours the millfoil contained over 8 mg/l., and after 10 months 58.8 mg/l. were found in the bottom sediments.

The reality of the danger in allowing herbicide to reach wetlands is strongly emphasized by Norris (1971). From field studies of forestry applications of phenoxy herbicides he found that residue persistence was not a problem in soils or streams, but cautioned that: "Applications of herbicides to marshy areas may result in high-level, long persistence of chemical residues in nearby streams. Special care must be taken to avoid treatment of such areas."

Although experimental toxicity data are available for many aquatic organisms, virtually nothing is known about the effects of herbicide drift into prairie wetlands. Indications are that even relatively small amounts of herbicides can recognizably alter the structure and functioning of wetland ecosystems. Walker (1971) has shown that 0.5 ppm of sodium endothall applied to a small pond selectively killed certain submerged plants and rechannelled energy flow through decomposer organisms. In response, a wide-range of invertebrate populations were significantly affected. Similarly, Walker (1962) demonstrated that 1 to 4 ppm of 2,4-D reduced a large number of benthic invertebrate populations within one week of application.

A vegetation study by Walker and Wehrhahn (1971) in Saskatchewan, although not dealing with herbicides, provides circumstantial evidence for the effects of herbicides in wetlands.

In this study, the relationships between variation in the vegetation and environment were investigated in relatively undisturbed, non-to slightly saline, shallow marsh habitat, all surrounded by large areas of wheatland. Of the many physical and chemical parameters studied they found that the water chemical conditions resulting from high rates of decomposition were the most important in determining the distribution of plant species. Although the authors suggest that the decomposition was a result of natural winter die-off, such conditions are a typical aquatic ecosystem response to phenoxy herbicides (Walker 1971; Smith and Isom 1971). The fact that all the wetlands studied were surrounded by wheatland indicates that they could be exposed to herbicides.

If plant species composition in prairie wetlands is being affected by herbicides, the response should be reflected throughout all trophic levels in the ecosystem. Work by Dwyer (1970) in Manitoba indicates that the whole biota of wetlands in agricultural and non-agricultural lands is recognizably different. Although both study areas were on the same soil type there were recognizable differences in the distribution and abundance of submerged plants. Populations of aquatic invertebrates followed a similar trend. In the agricultural wetlands free swimming invertebrates were dominant. Close to 80% of all invertebrates were amphipods, whereas in the non-agricultural lands this group made up less than 1%.

Since phenoxy herbicides are known to accumulate in pond sediments they could be selectively toxic to bottom organisms, and be instrumental in changing aquatic invertebrate fauna from balanced benthic and free swimming assemblages to a less diverse fauna with primarily free swimming populations. Further, because of the known ability of herbicides to persist in aquatic ecosystems, wetlands in the vicinity of croplands could be functioning as herbicide sinks.

The widespread use of herbicide formulation which are able to contaminate the environment may be taking modern agriculture far beyond its obvious level of environmental uniformity to the point where the productivity and diversity in portions of the natural environment is markedly reduced.

Document A

TABLE 57. The LC_{50} for various fish to 2,4-D.

Formulation	Fish Species	Exposure Time (hr)	LC_{50} (ppm)	Source
Butyl Ester	Harlequin fish	24	1.0	Alabaster, 1969
Oleic-1,-propylene diamine	Bluegill	24	4.0	Davis and Hughes, 1963
Butyl Ester	Bluegill	24	4.9	"
Butyl Ester	Bluegill	24	10	"
	Rainbow trout	24	250	Alabaster, 1956
Amine	Rainbow trout	24	250	Alabaster, 1969
Ethylhexy Ester	Lake Emerald shiner	24	280	Swabey and Schenk, 1963
Ethylhexy Ester	Lake Emerald shiner	24	620	"
Sodium Salt	Harlequin fish	24	1,160	Alabaster, 1969
Isopropyl	Bluegill	48	0.8	FWPCA, 1968
Propylene Glycol Butyl Ether Ester	Rainbow trout	48	0.96	"
	Rainbow trout	48	1.1	Bohmont, 1967
Butyl Ester	Bluegill	48	1.3	FWPCA, 1968
Mixed Butyl and Isopropyl Esters	Bluegill	48	1.5	"
Butoxyethanol Ester	Bluegill	48	2.1	"
	Bluegill	48	3.7	Bohmont, 1967

The minimum lethal concentrations (ppm) of 2,4-D which produced a kill of fish-food organisms exceeding 25 percent are the following: *Daphnia*, 0.2; *Eucypris*, 0.6; *Hyallella*, 0.6; *Palaemonetes*, 0.8; *Amphiagrion*, 3.0; *Pachydiplax* and *Tramea*, 4.5; *Culex*, *Aedes*, and *Anopheles*, 3.5; *Chironomus*, 1.0; *Physa*, 5.5; and *Helisoma*, 7.5 (Zischkale, 1952).

From. David Pimentel. 1971. Ecological Effects of Pesticides on Non-Target Organisms.

B. An Assessment of Blanket Herbicide Spraying on Rights-of-way.

Close to one million acres of land in Alberta are used as rights-of-way for roads, railroads, pipelines, utility transmission lines and irrigation projects (Oetting 1971). For the most part, this property is not owned by the utility agencies, but is a public resource. However, vegetation control on these lands is normally accomplished in the most economically advantageous way to the specific agency, with no consideration of the public. With certain exceptions, such seeding highway rights-of-way, the almost universally practiced method of vegetation control has been to blanket spray with herbicides. Blanket spraying is considered cheap, fast, and it has been strongly promoted by chemical manufacturers, spray companies, and federal and provincial departments of agriculture. As a result, large areas of highly visible public property have been repeatedly defoliated in the name of vegetation management.

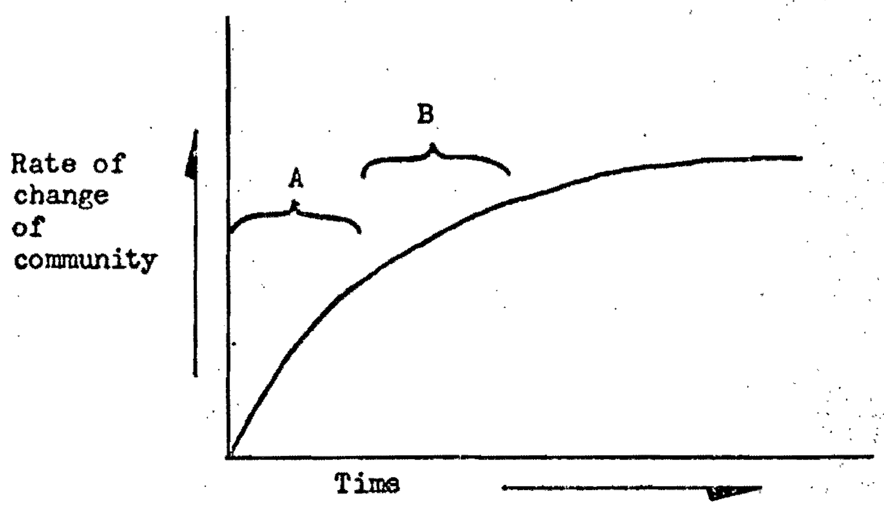
The Canadian public has been reacting strongly against these practices on the basis of aesthetic damage and the loss of valuable wildlife habitat. As a result of public pressure, New Brunswick and Nova Scotia have recently banned the use of herbicide on rights-of-way. In Alberta, the utility companies and government agencies are receiving a large number of complaints.

By and large, the herbicides presently used on right-of-way maintenance are not directly toxic to higher animals. Thus, the users of the chemicals say that they are not harming wildlife. However, in terms of food and cover requirements, the use of herbicides may be an extremely important limiting factor.

The objectives of vegetation control on rights-of-way are generally twofold. First, there is the requirement to remove or supress forms of vegetation which physically impair the function of the utility. In most cases trees are the offending plants. The second objective is weed control.

A basic knowledge of plant succession phenomena suggest that blanket spraying is ecologically unsound, and in fact has a builtin feed-back mechanism which increases the requirements for further spraying.

Rate of change curves for successional plant communities are usually convex, with changes occurring most rapidly at the beginning (Odum 1969).



Right-of-way plant communities should be managed to be in stage B of the curve, where the rate of change is slower, and shrubs provide stability and competition to weeds and trees. Instead, the rights-of-way are blanket sprayed so that they remain in stage A. This is a much simpler biotic community, with a high rate of change, offering little inherent stability or competition to weeds or trees.

It is said that denuding the offending brush of foliage brings on green grass growth. The ecological instabilities of this "green grass growth" is illustrated by the fact that blanket weed spraying has to be conducted annually to satisfy the criteria of the Weed Control and Field Services Branch.

Egler (1953) found that between the majority of trees and shrubs, the trees are more resistent to foliage sprays than the shrubs. Thus, by the time the trees are finally killed, the shrubs and desirable coarse herbs are practically annihilated. Blanket spraying in doses high enough to kill trees leaves only grass cover. There is little wildlife cover, and there is minimal competition to offending trees and weeds. Blanket spraying for weed control on these areas is required annually, and every 5 years or less for tree control.

Egler (1953) has demonstrated in the U.S. that desirable shrubs are part of the initial floristic composition. If blanket spraying is not done, and only the offending trees are spot treated, a shrub herb-grass community develops which is ecologically more stable. It provides excellent wildlife habitat, and offers competition to trees and weeds. As an aded bonus, less maintenance is required, so that it is more economical to the utility agencies.

It is rather ironic that manufacturers and promoters of agricultural chemicals are in a sense, managing ecosystems for their own economic gain. Knowingly or not, by promoting blanket herbicide spraying, they are catalyzing a biological feed-back mechanism based on the homeostatic properties of ecosystems to increase the requirement for their products. Thus, agricultural chemical companies are managing ecosystems for an economic return far more successfully than the wildlife profession is for a wildlife return.

The management of vegetation communities on rights-of-way should be based on sound ecological principles, not just economic ones. The communities should be managed to satisfy the primary users of the right-of-way, be aesthetically pleasing, and be of value to the environment as a whole.

C. An Ecological Evaluation of Picloram.

The Weed Control and Field Services Branch of the Alberta Department of Agriculture has been strongly recommending the use of picloram on wastelands, fencelines and native pastures. The widespread use of this chemical in Alberta could have very serious environmental implications.

The properties of picloram which are responsible for its successful control of perennial plants (persistence, high phytotoxicity and movement in water), preclude its widespread use in agricultural crops. For example, field experiments in Ohio showed that picloram applied to the soil at 2 lb/A nine months before sowing, did not affect corn or oats but caused 40 to 50% reduction in barley and killed alfalfa and soybeans (Herr. et al., 1966). Keys and Friesen (1968) mention that grain crops, notably wheat, are reduced in height and yield by as little as 0.75 oz./A of picloram.

Not only is persistence of picloram a concern at the site of application, but because of its high solubility in water, there is a danger of contaminating other crops through lateral groundwater movement or contamination of irrigation water.

Baur et al. (1970) treated several crop species with single doses of very low levels of picloram. The rates used were similar to concentrations found in runoff water adjacent to and up to 2 miles from a 15 acre watershed treated with 1.0 lb/A of picloram.

Depending on the plant species and concentration, picloram was found to be both growth promotive and inhibitory. There was a significant stimulation in dry weight of corn, sorghum, cotton, and soybean with treatments of 0.25 ppb, and cowpea at 1.0 ppb. However, there were significant reductions in the dry weight of corn, wheat and sorghum at 1,000 ppb; and at 100, 10, and 1.0 ppb for sunflower, cowpea and soybean respectively. Visual effects of picloram were noted on sunflowers at concentrations as low as 0.25 ppb. Bovey and Scifres (1971) concluded that residues of 10 ppb or more in irrigation water could severely affect the growth of some crop seedlings.

The majority of herbicides presently in use are employed for selective weed control in agricultural crops. Evaluations of the performance of such herbicides are primarily based on the efficiency of the chemical in controlling unwanted vegetation and the tolerance of the crops. However, cropland monocultures are, by fundamental ecological standards, simple and highly unstable ecosystems. Stability in the face of biological, edaphic and climatic stress is gained only through vast subsidies of energy and matter. Thus, agricultural herbicides are used and evaluated in ecosystems where the energy input required to keep environmental stresses to a minimum is often substantially greater than the energy yield realized in the food produced (Brown 1971).

In contrast, picloram is used in more natural ecosystems where the herbicide treatment is often the only material input to the system. It follows that an evaluation of the toxicity of picloram to vegetation must relate to the whole plant community within an ecosystem context.

Picloram containing herbicides are regarded as having considerable potential for the control of woody and herbaceous plants on overgrazed rangelands (Arnold and Santelmann 1966), (Scifres and Halifax, 1972), (Bovey and Scifres, 1971). The object is to simplify the plant community by killing brush and herbaceous vegetation and promoting more grass cover to sustain heavier grazing.

The inherent danger in simplifying natural ecosystems is the possible impairment of their stability. Whereas short-term yield is of prime importance in the artificially stabilized cropland systems, the concern in native rangelands is for long-term yield. The stability of a complex mixture of plant species may influence the average yield of grasslands over a longer period than is the case with simpler systems (Spedding, 1971). Further, a diverse rangeland flora is less susceptible to serious loss from adverse climatic conditions or insect damage (Vallentine 1971). Diversity gives more buffering against drought. A mixture of more or less drought resistent plant species, with appropriate seasonal growth patterns, may be more productive than any of the species grown alone (Spedding 1971). The ability of the plant community to withstand drought is of particular importance in western Canadian rangelands where there are frequently successive dry years which reduce the abundance and cover of the higher yielding grasses (Campbell et al., 1962).

There has been relatively little research done on the effects of picloram on desirable species in rangeland ecosystems (Scifres and Halifax 1972a). Not only are brush and forbs affected but detrimental effects have also been found on desired rangeland grasses.

Arnold and Santlemann (1966) treated established native range in Oklahoma with picloram at rates of up to 4 lb/A. All rates reduced forb production without reduction in desirable grasses. However, pre-emergent treatment with picloram at 0.75 lb/A prevented germination of side-oats grama (Bouteloua curtipendula), big bluestem (Andropogon gerardi), switchgrass (Panicum virgatum) and blue grama (Bouteloua gracillis) in the greenhouse, and decreased yields at the two and four leaf stages. All species became more resistent to picloram at advanced growth stages.

According to Bovey (1971) persistence of picloram in soil and the movement of picloram in water are not considered to be objectionable in non-croplands, but are thought to be necessary to arrest regrowth of woody species after initial defoliation by foliar sprays.

However, concern over soil residues following picloram application to rangelands and the sensitivity of some grass seedlings led to a study by Scifres and Halifax (1972a) on root production of seedling grasses in picloram treated soil. With picloram residues in the soil, it was found that foliage of switchgrass was developed at the expense of root production. There was no differential inhibition of side oats grama roots and shoots, although root penetration was inhibited when picloram was present in the surface inch of soil. Responses of these two species of grass seedlings to varying concentrations of picloram in surface soil are shown in Figure 1. Scifres and Halifax concluded that picloram applied just prior to or following range seedling, or applied to badly depleted rangeland, could further complicate the rate of reseeding establishment.

In a subsequent experiment Scifres and Halifax (1972b) found that picloram differentially regulated post-germination growth of range grass seedlings. Buffalo grass (Buchloe dactyloides), which was the most resistent of five grasses studied, showed significant root length reduction at 125 ppb of picloram. Again, with all species, picloram had a greater effect on roots than shoots.

The fact that roots are differentially affected by picloram could seriously impair the resistence of rangeland ecosystems to grazing and drought.

It is generally concluded from the available toxicological data that the use of picloram presents no hazard to humans, livestock or wildlife (McCollister and Leng, 1969; Bovey and Scifres, 1971). While this is true from the point of view of the danger of direct toxicity, it cannot be assumed that there are no hazards to wildlife when habitat requirements are considered.

Wildlife species cannot be considered singularly as they are in toxicity experiments. In their natural environment, they occur as a number of co-existing populations. The status of each population cannot be simply defined, but involves an integration of many parts of the ecosystem. Habitat requirements such as food and cover are not constant, but vary both spatially and seasonally. Thus, diversity of habitat is of major importance not only to the number of species, but also to the well-being of individual populations.

It is recognized that herbicide use can benefit certain wildlife populations when it promotes heterogeneity in the landscape (Springer 1957). Conversely, reducing habitat diversity can be detrimental.

The use of picloram in non-croplands may present a danger to wildlife because of the phytotoxic affect on such a broad range of plant species. On native rangelands both livestock and wildlife feed extensively on browse and forbs as well as grasses (Vallentine 1971). In fact, browse and forbs are considered to be an asset to a range because they add variety to the diet and enhance the nutritional intake of both livestock and wildlife (Sampson 1952). A well balanced rangel nd is composed of browse and forb species which far outnumber the native grasses (Sampson 1952). Simplifying the system to one composed primarily of grasses impairs the variety of animal species dependent on the native vegetation. Diversity of plants is also important in providing a variety of plants for the changing seasonal needs of the animals. Herbaceous plants are not only of direct nutritional value as forage, but the killing of legumes can result in the removal of an important nitrogen source to the plant community (Spedding 1971).

The application of picloram to non-croplands can have a greater impact on the vegetative community than the reduction of woody or herbaceous species. The differential effects of picloram on grass species, especially at seedling stages, may alter grass competition and further reduce the diversity of the community. The simplification of the ecosystem may lower its resistence to environmental stresses such as drought, heavy grazing, insect damage and disease. Moreover, the fact that picloram can damage the roots of grasses and retard seedling growth could increase the susceptibility to such stresses.

Similar problems can occur in the treatment of rights-of-way. In agricultural areas or other types of land where there is little cover, treatment with herbicides may be detrimental to wildlife (Springer 1957). Picloram could be of particular significance in this respect because of the broad range of species killed. Also, because of its persistence and mobility in water picloram's effects can be felt over an area much wider than the original site of application. Scotter (1971) found that an application of Tordon 101 along the 20-foot wide international boundary between Waterton Lakes-Glacier International Peace Park killed trees up to 150 feet from that line as a result of spray drift. Also, phytotoxic effects were detected more than a hundred feet down slopes. It should be noted that in this operation a particulating agent was used to minimize drift.

Although picloram is not highly toxic to animals it has a high degree of toxicity to a wide variety of plants. Its use can fundamentally change the structure of plant communities, and as a result can be very detrimental to a wide variety of animal life.

It is essential that the use of picloram on non-cropland vegetation be evaluated, not only from short-term criteria, but on the long-term ability of ecosystems to maintain stability in the face of adverse environmental conditions and to support both domestic and native animal life.

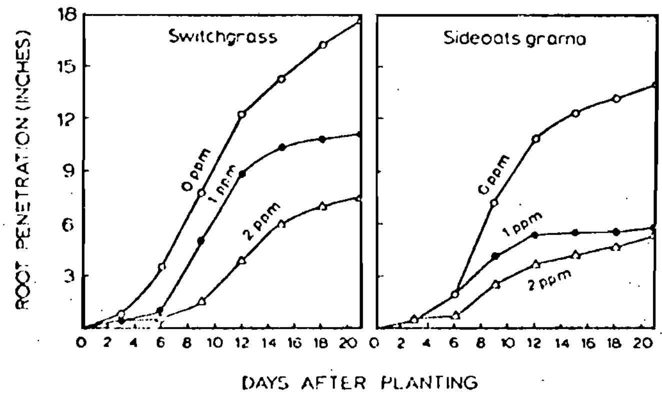


FIG. 1. Average root penetration of switchgrass and sideoats grama seedlings when the surface inch of soil contained picloram at 0, 1 or 2 ppm.

Figure 1. From C.J. Scifres and J.C. Halifax. J. Range Mgmt. 25:44-46. 1972.

Literature Cited

Anonymous, 1972. Weed-O-Gram. Weed Control and Field Services Branch, Alberta Dept. Agr. Vol. 4, No. 9.

Arnold, W. R., and P.W. Santelmann. 1966. The response of native grasses and forbs to picloram. Weeds. 14:74-76.

Baur, J.R., R. W. Bovey and C. R. Benedict. 1970. Effect of picloram on growth and protein levels in herbaceous plants. Agron. J. 62:627-630.

Bovey, R. W. 1971. Hormone-like herbicides in weed control. Econ. Bot. 25:385-400.

__________, and C. J. Scifres. 1971. Residual characteristics of picloram in grassland ecosystems. Texas Agr. Exp. Sta. Bull., B-1111, pp. 24.

Brown, L. R. 1971. Human food production as a process in the biosphere. Scientific American. 223:161-170.

Campbell, J. B., R. W. Lodge, A. Johnston and S. Smoliak. 1962. Range Management of grasslands and adjacent parklands in the prairie provinces. C.D.A. Pub. 1133, pp. 32.

DeMarco, J., J. M. Symons, and G. G. Robeck. 1967. Behaviour of synthetic organics in stratified impoundments. J. Amer. Water Works Assoc. 59:965.

Dwyer, T. J. 1970. Waterfowl breeding habitat in agricultural and non-agricultural land in Manitoba. J. Wildl. Mgmt. 34:130-136.

Egler, F. E. 1953. Our disregarded right-of-ways - ten million unused wildlife acres. Trans. 18th N. Am. W. Conf. pp. 147-158.

Foy, C. L. and S. W. Bingham. 1970. Some research approaches toward minimizing herbicidal residues in the environment. Residue Reviews 29:105-135.

Grover, R., J. Maybank and K. Yoshida. 1972. Droplet and vapour drift from butyl ester and dimethylamine salt of 2,4-D. Weed Science. 20:320-324.

Hay, J. R. and R. Grover. 1967. Recovery of 2,4-D from atmosphere. 21st Annual meeting, Nat. Weed Comm. (Canada), West. Sect., p. 43.

Hemmett, R. B. 1972. The biodegradation kinetics of selected phenoxyacetic acid herbidides and phenols by aquatic microorganisms. PhD Thesis, Rutger Univ., 136 pp.

Herr, D. E., E.W. Stroube and D. A. Ray. 1966. Effect of Tordon residues on agronomic crops. Down to Earth. 21:17-18.

Kenaga, E. E. 1969. Tordon herbicides - Evaluation of safety to fish and birds. Down to Earth. 25:5-9.

Keys, C.H. and H. A. Friesen. 1968. Persistence of picloram activity in soil. Weeds. 16:341-343.

Maybank, J. and K. Yoshida. 1969. Delineation of herbicide drift hazards on the Canadian prairies. Trans. ASAE, 12:759-762.

McCollister, D. D. and M. L. Leng. 1969. Toxicology of picloram and safety evaluation of Tordon herbicides. Down to Earth. 25:5-10.

Norris, L. A. 1971. Chemical brush control: assessing the hazard. J. Forestry, 69:716-720.

Odum, E. P. 1969. The strategy of ecosystem development. Science, 164:262-270.

Oetting, R. B. 1971. Right-of-way resources of the prairie provinces. Blue Jay. 179-183.

Pimentel, D. 1971. Ecologicak effects of pesticides on non-target species. Executive Office of the President. Office of Science and Technology. pp. 220.

Sampson, A. W. 1952. Range Management-Principles and Practices. John Wiley & Sons. N.Y. pp. 570.

Scifres, C. J. and J. C. Halifax. 1972a Root production of seedling grasses in soil containing picloram. J. Range Mgmt. 25:44-46.

______________________. 1972b. Development of range seedlings germinated in picloram. Weed Science. 20:341-344.

Scotter, G. W. 1971. Effects of Tordon 101 (4-amino-3,5,6-trichloropicolinic acid) on vegetation along the international boundary between Waterton Lakes-Glacier International Peace Park. Unpub. Manuscript Report. Canadian Wildlife Service, Edmonton. pp. 15.

Sheets, T. J. and C. I. Harris. 1965. Herbicide residues in soils and their vital toxicities to crops grown in rotation. Residue Reviews, 11.

Smith, G. E. and B. G. Isom. 1971. Investigation of effects of large scale applications of 2,4-D on aquatic fauna and water quality. Pesticides Monit. J. 1:16-21.

Spedding, C. R. W. 1971. Grassland Ecology. Clarendon Press, Oxford. pp. 221.

Springer, P. F. 1957. Effects of herbicides and fungicides on wildlife. North Carolina Pesticide Manual: 87-106.

Vallentine, J. F. 1971. Range Development and Improvements. Bringham Young University Press, Provo, Utah. pp. 516.

Walker, B. H. and C. F. Wehrhahn. 1971. Relationships between derived vegetation gradients and measured environmental variables in Saskatchewan wetlands. Ecology 52:85-95.

Walker, C. R. 1971. Toxicological effects of herbicides and weed control on fish and other organisms in the aquatic ecosystem. Proc. Eur. Weed Res. Coun. 3rd Int. Symp. Aquatic Weeds 1971. 119-127.

__________. 1962. Toxicological effects of herbicides on the fish environment. Ann. Air Water Poll. Conf., Proc. 8:17-34.