Regional Forester (Reviewing Officer)
Pacific Northwest Regional Office
Attn: 1570 Objections
P.O. Box 3623 Portland, OR 97208-3623
OBJECTIONS TO OCHOCO WILD HORSE HERD MANAGEMENT PLAN
Decision Notice, Finding of No Significant Impact, Environmental Assessment
From: Melinda Kestler

January 3, 2021

To Whom It May Concern:

Please find the following emails with attachments:

Appendix A Statement that all documents submitted still remain standing

List of Cited References, PDF

Attachments in Documents:

Follow up for Friday's winter range (9-10-18 email to FS re: gps map coordinates), PDF

ONF Scoping Letter, PDF

Sheep Permittee letter (Gordon Clark), PDF

Cothran/Gibson Conversation, PDF

Appendix B Strategic Research Plan Wild Horse and Burro Management, 2003 revised 2005, The BLM Wild Horse and Burro Program U.S. Department of Interior, Pgs. 1-45, PDF.

The 2020 Five Domains Model: Including Human–Animal Interactions in Assessments of Animal Welfare, Pgs. 1-24, PDF.

Seasonal Variation in Habitat Selection by Free-Ranging Feral Horses Within Alberta's Forest Reserve, Pgs. 428–437, PDF.

Management Implications of the Ecology of free-roaming horses in Semi-arid Ecosystems of the western United States, Erik Beever Pgs. 887-895, PDF.

Background for NEPA Reviewers: Grazing on Federal Lands, U.S. Environmental Protection Agency, Pg. 1-39, PDF.

Appendix C Comparative Reproductive Biology of North American Feral Horses, Jay F. Kirkpatrick, PhD, and John W. Turner, Jr., PhD; Equine Veterinary Science, Volume 6, Number 5, pages 224-230, PDF.

Habitat Evaluation: Guidance for the Review of Environmental Impact Assessment Documents U.S. Environmental Protection Agency Pgs. 1-129, PDF.

Biological and Social Issues related to Confinement of Wild Ungulates, The Wildlife Society, Pgs. 1-29, PDF.

OCHOCO WILD HORSE HERD MANAGEMENT PLAN OBJECTIONS

Melinda Kestler

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OCHOCO WILD HORSE HERD MANAGEMENT PLAN:

Melinda Kestler

Statement: All documents still remain standing.

Cited References:

Strategic Research Plan Wild Horse and Burro Management, 2003 revised 2005, The Bureau of Land Management, Wild Horse and Burro Program U.S. Department of Interior, Pgs. 1-45.

The 2020 Five Domains Model: Including Human–Animal Interactions in Assessments of Animal Welfare David J. Mellor 1,*, Ngaio J. Beausoleil 1 , Katherine E. Littlewood 1 , Andrew N. McLean 2 , Paul D. McGreevy 3 , Bidda Jones 3,4 and Cristina Wilkins 5, Pgs. 1-24

Using Science to Improve the BLM Wild Horse and Burro Program: A Way Forward (2013), Pgs.1-459

Seasonal Variation in Habitat Selection by Free-Ranging Feral Horses Within Alberta's Forest Reserve Tisa L. Girard,1 Edward W. Bork,2 Scott E. Nielsen, 3 and Mike J. Alexander4, Rangeland Ecol Manage 66: Pgs. 428–437 | July 2013 | DOI: 10.2111/REM-D-12-00081.1

Management Implications of the Ecology of free-roaming horses in Semi-arid Ecosystems of the western United States, Erik Beever Pgs. 887-895.

BACKGROUND FOR NEPA REVIEWERS: GRAZING ON FEDERAL LANDS, U.S. Environmental Protection Agency, Pg. 1-39.

Comparative Reproductive Biology of North American Feral Horses, Jay F. Kirkpatrick, PhD, and John W. Turner, Jr., PhD; Equine Veterinary Science, Volume 6, Number 5, pages 224-230

Habitat Evaluation: Guidance for the Review of Environmental Impact Assessment Documents U.S. Environmental Protection Agency Pgs. 1-129

Biological and Social Issues related to Confinement of Wild Ungulates, The Wildlife Society, Pgs. 1-29.

Exhibits:

9-10-18 -email letter to FS RE; map coordinates for fly overs

2017 scoping letter

Sheep permittee letter

From:	M G
To:	Gayle Hunt
Cc:	M G
Subject:	FW: Follow-up for Friday"s winter range information
Date:	Monday, November 30, 2020 11:35:18 AM
Attachments:	5CF8287710FE49B0835FAF12D134FD45.png

Sent from Mail for Windows 10

From: wildhorse.ccnr.pac@gmail.com
Sent: Mon 9/10/2018 4:15 PM
Cc: M G
Subject: Fw: Follow-up for Friday's winter range information

From: wildhorse.ccnr.pac@gmail.com
Sent: Mon 9/10/2018 4:15 PM
Subject: Re: Follow-up for Friday's winter range information

Hi Tory,
I would be happy to bring my small GPS in for you to try to download the wild horse sightings from. However, my aviation GPS has all my wildlife GPS points from my flights in Alaska and Canada. With that being said, I am not willing to take the chance of losing the information or something happening to my equipment as it is an expensive instrument. I can give you a list of the points with the years if that would work for you.

Just let me know.

Thanks,
Mel

From: Kurtz, Tory L -FS <tlkurtz@fs.fed.us>
Sent: Monday, September 10, 2018 12:59 PM
To: Gayle Hunt (gdhunt4@gmail.com) <gdhunt4@gmail.com>; circlegranch@live.com <circlegranch@live.com>
Subject: Follow-up for Friday's winter range information

Hi Gayle & Mel!

Thank you both so much for taking time to come in and provide your data and discussions on winter range for the wild horses, I really appreciate it! Just to follow-up, like I was sharing, the most helpful data for me would be to get the GPS points that you collected from the flights and to have them separated out by years and whether it was horses or trails. I understand Mel that you have that data and would be happy to help you download it off your GPS anytime. If at all possible, it would be great to get this additional info this week and I can make myself available to work around your schedule.

Let me know if you have any questions and if there is a good time this week for you.

Thank you!

Tory Kurtz
Rangeland Management Specialist

Forest Service
Lookout Mtn RD/Crooked River NG

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f: 541-416-6695
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3160 NE 3rd Street
Prineville, OR 97754
www.fs.fed.us

Caring for the land and serving people

Ochoco Wild and Free Roaming Herd (Horse) Management Plan Revision Project
c/o Marcy Anderson
Lookout Mountain Ranger District
3160 NE Third Street
Prineville, OR 97754

This letter is in response to the Ochoco Wild Horse Herd Management scoping letter of June 19, 2017 regarding the proposed update of the 1975 Ochoco Wild Horse Herd Management Plan. RE: File Code 1950

I moved to Prineville specifically because of the Wild Horses of the Ochocos. They are a very unique herd of wild timber horses with their own distinct conformation which reveals their true wildness. The longer I live here, the more people I find treasure the horses and consider them a valuable resource for the community; whether it be for photographing them for pleasure and profit or making a camping trip to come see them which ultimately fuels our local economy. There is an historical and cultural aspect that must be included in any resource decisions on this herd. This community has grown up with these horses; they are part of their past and citizens want to preserve them for generations to come.

First, I would like to make a statement that the scoping letter has given false and misleading information to the public. This is a violation of the NEPA process. This can only "lead or steer" the public to certain opinions as the situation of the herd and territory. The Ochoco National Forest, (ONF) has disclosed information with a preconceived statement that the public can only conclude: 1. there are too many horses, 2. many horses perished because there is not enough winter forage for them, 3. we must take down the numbers to save the rest of the herd, 4. the only reason why any survived is because they left the territory in search of food on private land. The scoping statements can only lead the public to conclude the same alternative as the ONF has already decided on.

The existing on the ground conditions, that have been put out to the public, did not include identified measurement indicators. The public expects and relies on ONF to be transparent and provide scientific and factual data. The annual census had not taken place yet, so no actual numbers were of record, this was totally presumptive data or preconceived to publish to the public. There is no factual data that supports your statement(s).

The ONF staff has gone back through the Environmental Assessment for the 1975 plan and selectively chosen speculative scenarios presented by the author of the EA that "the situation ***would probably*** occur where stallions would split off from the existing herds and establish new ranges outside the existing feral horse territory", as was his reasoning why the herd would leave their territory. This is in direct contradiction to his earlier statement, "The forage figures indicate that enough feed is available . . . to support an additional number of horses." This author is not a wild horse expert and had no knowledge of band structure of wild horses nor could he provide any supporting data back in 1975 to make such statements.

If the facts presented by ONF were true about horses leaving the territory due to increase in herd size and the lack of available winter forage, then why did the annual census tell a whole different story? The herd is still in the territory and intact, with minimal deaths due to snow load and conditions on the forest floor, contrary to the scoping letter. The public demands and expects scientific facts not fiction, to base their responses on. If allegations and complaints of the Big Summit wild horses being out of their territory have been alleged, the public expects confirmation of fact, that they truly are part of the Ochoco herd. Having many conversations about "feral" horses with the local ranchers, it is the consensus that the horses that are running outside of the forest are, "saddle bred" horses that other ranchers have turned loose.

NEPA requires all federal agencies to consider the impacts of their action on the environment, which the wild horses are truly apart of the ecosystem. They have existed there at a very minimum for centuries. Research based on long time, local residents' statements clearly shows the dates and places of existence for the wild horses was not thoroughly investigated by the EA author that the ONF is quoting. As for riparian damage caused by horses, we have statements from locals, "herders move the sheep down the creeks and streams".

The purpose and need statement is the most important section of the environmental document, which is orchestrated based on the background, it establishes the reason why the ONF is proposing the project. In addition, the purpose and need statement frames and justifies the expected outcome of the public expenditure and allows or gives reason for the decisions to be defensible; it is where the problem should be articulated to the IDT team as well as the public. The phrase garbage in garbage out is an excellent way to put it. The footprint to this NEPA process is flawed in the framing stage; it is very clear where the ONF is taking any analysis by the IDT for the outcome that was cast as the problems.

The ONF has narrowed down an infinite range of alternatives to a select few that will fill the "underlying need" as has been put forth. The purpose of this proposal will become the decision factor the ONF will use to support the final selection of one of the alternatives. The ONF is using the purpose and need to set the scope of what alternative action will be considered. Under 40 C.F.R. §1502.2 (g) (requiring that NEPA review "shall serve as the means of assessing the environmental impact of proposed agency actions, rather than justifying decisions already made"); id. § 1502.5 (NEPA review "will not be used to rationalize or justify decisions already made").

Response to Proposed Action;

1. Per the scoping letter, "The AML analysis will calculate the winter forage available for horses and allocate the forage for maintenance of healthy horses", there is no science more powerful and accurate than the actual number of horses that made it through the winter given the conditions that they had to endure. There are no errors in this data calculation derived by nature itself. It wasn't a matter of forage but a matter of the energy it would take a horse to reach it. As I stated before, the wild horse herd made it through intact with minimal deaths. This proves the tenacity and resilience of the herd – truly survival of the fittest that made it through a drought year of forage. Data on the 2017 forage must be included in the trend line.

Analysis must be done on the predators of the forest and how it will impact the wild horse herd. We now have large numbers of cats, a growing number of wolves and bears are now present. Miners in the wild horse territory have observed cats tracking the herd as well as a wolf during the foaling season.

The statement that "horses above the identified AML range would be considered excess animals", is alarming in the notion that the FS is planning on removing any horses above the 1975 AML. There are several news releases and statements which have informed the public that the old plan is out of date and a new plan is needed before any action can be taken. The EA would be required by NEPA law to be completed before any removal of wild horses would be legal. The ONF is the responsible party that has amended the AML by their inaction of following the recommended AML of 1975 or all recommendations from that document.

2. Any decrease in the WHT would have to be proven on the ownership and how this mistake could have happened and why is it being corrected now. There is no mention or information on this in the 1975 EA. If this can be corrected administratively, requests have been made to ONF to correct the WHT and expand the territory to the areas where the horses were located during the 1975 EA, to bring it into compliance with the Wild Horses and Burros Act as to the location where they existed. Documentation on the locations of the other areas that should have been included in the WHT are noted in the 1975 EA. It

was previously stated by ONF that it would take an act of congress to change the territory boundaries.

3. My family has been involved with horses for over 50 years; breeding, raising, training. Having knowledge in equine breeding, based on equine genetic DNA, the ONF does not have the qualified staff to understand the implications of cross breeding. A huge array of considerations must be made and an understanding of what you will ultimately be producing. This herd would be genetic test rats for a trial and error breed program, which is not in the best welfare of the wild horses. The specific DNA of this herd must be preserved if not for the historical marker that they carry but for the characteristic traits that they possess. ONF admittedly also does not have the budget to properly monitor. "All *management activities* shall be at the minimal feasible level", which an augmentation program certainly couldn't be carried out with "minimal management".

All equine geneticists have the same opinion on herd size to achieve genetic viability including, Gus Cothran of Texas A & M (referred to in the scoping letter) of 150 – 200 horses. At one point we reached 152 horses, and the conformation of the offspring proves their data. Below that, the herd has been bottlenecking and the deformation has manifested in certain bands.

4. I agree with fertility control using PZP. The Central Oregon Wild Horse Coalition has members who are willing to be trained at the Billings, Montana facility to administer PZP at our own expense. In addition, I am also willing to obtain my Oregon applicators license under the EPA rules.

5. The large WHT working group worked diligently for over a year and a half on the emergency plan, rescue, with only minimal parts accepted by the ONF. Yes, I agree we need a plan but most importantly we need the people with the certificates and experience. The COWHC members are technical, large animal certified experienced rescuers, with the necessary equipment for equine rescues. I am a certified wild animal rescuer with rehabilitation training which includes body condition scoring. The COWHC is a local group that can assemble members at a moment's notice – in any rescue, time is of the essence!

6.) Yes, programs need to be implemented. Once again, the COWHC has taken on the adoption responsibilities of the horses that we have been able to rescue. The Coalition has local facilities to take these horses, gentle down and find them good homes.

Melinda Kestler
[redacted]
[redacted] [redacted]

HAYCREEK RANCH _

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April 11, 2016

Dear Mr. Peña and Ms. Li,

First, thank you very much for your response to my February 25 letter. It was a very good letter.

I especially appreciate Ms. Li reviewing some of the issues I brought up. I think before this whole horse mess the Ochoco National Forest Staff have created is cleared up you are going to really appreciate the point that ***"the Forest Service does not have the same restrictions on appropriations as the BLM"***. I urge you to keep this foremost in your mind as the law is going to help you! It is a well written, easy to understand law. A variety of solutions to the problem are well spelled out in very plain English.
This letter does not need a response. I am sending it for the purpose of helping you and the people working under you make decisions that will resolve the Big Summit wild horse herd problem. It should be considered an addition to my earlier letters. I apologize for the length of this letter but it was required as your problem is big and complex.

Ms. Li, you can follow what I am saying and possibly make a determination from the legal point of view. That might be helpful for Mr. Peña.

From your letter it is obvious you have been misled as to the seriousness of the problems you face and the complexity that is involved.

I would also note you are strictly governed by the law passed by the Congress and signed by the President. The main law is the 1971 act and its amendments the last of which was added in 2005. The Forest Service has taken the position that they are allowed to "interpret" the law. That is reflected in the Forest Service Manual and probably some other Forest Service rules. There is also the BLM Handbook and other BLM rules your local Ochoco staff is relying on or quoting. Furthermore, there is a fair body of Case Law involved. You also have Mr. Barry Imler, the Forest Service Rangeland Program Manager, who can probably make further interpretations or decisions. As a warning a few members of your Ochoco staff have a bad reputation for more or less exerting their power arbitrarily and at the same time making up their own law or rules. In sum, they could cause you problems. Furthermore, this horse issue is probably going to be testing interpretations of the law and Forest Service policy. Therefore, I think it would be helpful for you to do serious legal research in advance and make your decisions very transparent so we stakeholders know you are following the law and rules. This process does not belong in the court system and you alone can avoid this problem. You have a lot of latitude or power but there is a limit.

Earlier I have made suggestions on how to avoid litigation with the very litigious national wild horse organizations. You do not seem to care one way or another. Since they always lose I guess this is not important to you. I will drop this issue in the future. I would remind you, however, that litigation can be a very successful delaying tactic.

My viewpoint and expertise is that of a livestock rancher and farmer. I own a 52,500 acre fee simple ranch. I have over 700 acres of irrigated farmland. I have about 900 cows and over 4,000 sheep. I am an expert in all the tasks required to solve your horse problem. This includes fabrication, electronics (that you are unaware you need at this time), capture, identification, record keeping, processing, sampling, and medications. I am very experienced in the safety issues involved – and that is important. I am also very experienced in expediting a complex technical problem using unskilled labor. I might also add that my entire irrigation system is run from a cell phone using technology I built myself. This is similar to a system you will need to solve your problem.

On the other side I want to stand up for all of you, and especially the Ochoco staff. None of you trained for this stuff. None of you took your jobs expecting to be thrust into this type of work. While some of you are excellent administrators, it is very difficult to effectively manage something you really do not understand – especially if your staff is also inexperienced.

Besides inexperience there is another management problem. Your Ochoco staff is trying to do everything internally under the influence of what you called in your letter your “partner” – the Central Oregon Wild Horse Coalition (COWHC).

While on the subject of the COWHC, I would like to make an observation. When you are in business and you have a partner who is causing the business to lose money you do everything you can to end the partnership. Prior to Stacey arriving and shaking things up I would say the COWHC was running your wild horse program. As I noted in my earlier letter “current or past radical Wild Horse Advocates embedded in your staff or exerting significant influence in policy”. My summation is that they purposely got you in the mess you are now in. They know exactly what they are doing and are familiar with *law, case law, and the choice of "experts" to manage their agenda. Now they want an expansion of territory and a significantly increased AML. Their main pitch now will probably be based on inbreeding, as the herd has an inbreeding problem. My strong suggestion is that you reevaluate the partnership for they are far too expensive and are creating controversy. As time goes on you will finally realize how expensive they are. To date it appears you are clueless! Later in this letter I will cover this a lot more.*

In my earlier letter regarding your Ochoco staff I also noted "their present courses of action or methods have been tried elsewhere and failed". This is a very good description of what is going on now – especially with your useless meeting process. It is crystal clear what needs to be done so why have three years of meetings while your herd expands at up to 25% per year? You needed to start acting YESTERDAY!
To get you warmed up I will bring up a few points to explain the extent of the problems you are facing:

1. The Burns BLM facility was full in 2011 and they refused to take horses from you at that point. The 2013 figure you quoted is the official date the BLM shut down. The 2011 date fits what we noticed as far as forage decrease and resource damage increase goes. Since 2011 your herd should have at least doubled – but no one knows exactly where all of your horses are now located. You do not know exactly how many horses you have. (I believe in the last few months your internal count jumped from about 115 to over 160? I cannot confirm this. That is not even beginning to count your horses that are not where they are supposed to be.)

2. Here is the local adoption situation. The Burns BLM Facility is now very full of the most desirable wild horse genetics in the United States. The Burns horses from the Steens are in part traced to the original Spanish Conquistadors. They are hand picked for adoption. At this point when they capture horses to put in the facility they sort them for quality and return the less desirable horses to the wild. Even with that selection their adoption rate is approaching zero. The nearby Warm Spring Indian Reservation near Madras also has a large wild horse herd and an adoption program. It is also not doing well. They are evidently shipping excess horses out of the country for slaughter. I have not followed your Murder's Creek problem but it is probably a factor. The note in your letter that "adoption" is one of the tools you are relying on is straight from COWHC and is not realistic. They know it but your staff in the Ochocos evidently does not or they are not giving you straight answers. Furthermore, your herd has some very undesirable genetics, as locals believe it is made up from recently abandoned horses. It has some obvious inbreeding issues. The main reason for the lack of adoption demand was the significant increase in the cost of hay, veterinarians, and the economy. There is an excess of domestic horses. Abandonment has become so prevalent that it is now a felony in Oregon. (Your "wild horse" herd even had some metal horseshoes and brands.)

3. When considering your horse problem keep in mind the BLM claims the average life of a wild horse is 16-18 years. This means that theoretically a roughly 7%

reduction in herd size each year if no new colts are born. Mares between 5 and 10 years of age are the most prolific. All studies come up with a 15% to 25% herd size increase each year. The BLM now feels 25% is the more accurate figure. Compare all of that to the plan stated in your letter to me. Also factor in that you do not have an accurate count plus you might be responsible for an awful lot more horses than your people are reporting to you.

4. At your first public meeting for the new management plan I was the only grazing permit holder present. There was only one small landowner. It appeared the rest of the room was filled with COWHC members or other horse people along with your staff. (The meeting was adequately advertised.) That is when your staff announced that an increase in the AML was being considered. (Stacey clearly stated at the meeting that an increase in territory was not going to be considered. Your letter says it is or you might have meant the boundary line would be changed but the overall acreage would be the same? The 1971 law clearly says you cannot increase the size. There is Case Law on this subject. I believe it is regarding a true “administrative error”, but the law is very clear wording so you may be exceeding your authority? This is clearly an issue for Ms. Li.) The word got out and the extent of the mismanagement problem is starting to surface and spread. At the same time I notice a clear cover up starting. It probably is extending to you. I want to make it very clear that Stacey is not a part of this cover up problem. It is more or less coming from the people within the staff that have been there as your problem was created.

I want to go over the history of the Ochoco grazing and the wild horse herd so you can get an idea what the modern Forest Service is in the process of dismantling over a relatively short period of years.

I have a book written by the Forest Service person who laid out the Ochoco grazing allotments we use. He did it on horseback. The year was 1908. The book was written about 1964. This means the grazing allotments you will probably be shutting down soon are over 100 years old.

Sometime just before or after World War II there were some sheep processing corrals and a scale built adjoining the wild horse’s designated area. This means my two affected allotments have been used continuously for sheep for 65 or more years. I removed the improvements around 2000 as we now use portable equipment.
My oldest living information source has been continuously in the area since 1959. He and the late Les Schwab tire magnate each own a large meadow inside of the Ochoco National Forest. (Big Summit Prairie. It is very clear on your maps of the area. It also adjoins the key Reservoir Sheep Allotment and the wild horse designated area. Ms. Li, he was also a key witness in a jury trial I am going to mention later in this letter.

Needless to say he is a very powerful witness as you will learn later.) There are evidently others who have been in the area for a very long time. (This should also be of interest Ms. Li.)

About 1970 the ranch I own took over three sheep allotments in the Ochocos. I purchased the ranch in 1993 and have used the allotments continuously since 1994. In

the 1990's the only other sheep allotment in the Ochocos was merged into one of the sheep allotments I use that is far from the horse area.
The key date for the horse issue was 1975. That is when the AML was set at 55 to 65 horses. It was evidently controversial at the time. Locals considered it too high and the Forest Service made a number of promises. I understand there are old newspaper clippings about the event around. That is also when the boundary was set.

Note you might look at a number of other Forest Service or BLM AML's. You will note the horse per acre is quite similar to the above number. Maybe the National Academy of Science who managed the above number method was really not as dumb as your local staff seems to think they were. You will also note your Ochoco herd will probably be the only one in the United States to actually increase their AML plus have the highest horse to acre ratio in the United States.

There is some controversy about the 1975 event.

The COWHC stated in the first public meeting that the 1975 boundary was an "administrative error" and should have been larger. Ms. Li will note case law clearly does not allow such a claim. The COWHC is trying to make some of the area outside of the legal boundary where a significant number of horses are now residing legal. Their goal is quite transparent.

For what it is worth at the present time the boundary is not important. The entire "legal" boundary is full of horses. There is an overflow outside of the boundary.

There is also evidently a claim being made by the COWHC and some of your older staff that there were other wild horse herds on private property during the 1975 period that were not from the Big Summit herd or they were from the Big Summit herd but legally adopted from the Forest Service. I was not around and neither was your staff involved or probably even COWHC members. This claim does not make sense for a number of reasons. This all allegedly took place on private property that does not allow trespass. Ms. Li, if I were in your position I would put the brakes on this fast. For the most part you would be up against wealthy, large landowners with lots of credible witnesses. They would probably have excellent legal representation. My witness mentioned above would be a key one and he is accessible if you wish to interview him. Simply put this is a bogus claim. The Forest Service has been breeding horses and feeding them onto private property.

I would note that during severe winters or periods of forage shortage it would be natural for horses to migrate to lower elevations or areas where there is more forage.

The other issue regarding the 1975 number is inbreeding. Without an effective management plan you cannot maintain a herd of that size without encountering a serious inbreeding problem. In hindsight the whole 1975 idea of a 55 to 65 number, unmanaged horse herd could probably be classified as a mistake.

I will cover inbreeding extensively later in this letter.
You might also note from the current Manuals this low number problem and the associated inbreeding problems involved must have surfaced other places. That is why you are supposed to eliminate the herd.

We also know that unwanted horses have been abandoned into the Big Summit herd – probably continuously since 1975. I can verify this has been happening to some degree since about 1994. This probably helped the inbreeding problem.

When I started using the Reservoir and Canyon Creek sheep allotments in 1994 there were very few wild horses around and very little manure signs. The Reservoir Allotment, the main part of the authorized wild horse range, had the best forage of the three allotments I use. That is where we always put our twin lamb bands as it had the best forage.

We never saw horses in parts of the Canyon Creek allotment they are not supposed to be in.

A few years ago I noticed a significant increase in the number of horses and resource damage in riparian areas. I notified our District Ranger by letter and got no response. Since that time we have noticed the rate of increase of horse numbers was increasing at an ever more rapid rate. (This probably ties directly to the research showing a 25% per year increase in herd size.) The District Ranger was again notified with no response. The situation was mentioned in numerous meeting with staff as well as the District Ranger. They all seemed aware of the problem. Nothing was being done.

Last summer we evidently hit the limit. It was a mild drought year with a very warm late spring period. The adjoining Big Summit Prairie cattle grazing was started early but the overall numbers or AUM's were normal for the season. Their range came out in good shape. The Reservoir Allotment, our best forage Allotment, ran out of forage. The Reservoir riparian areas were trashed. We noticed a spreading of horses into the Canyon Creek Allotment. There were even horses on adjoining Cattle Allotments.
Very recently three things of significance have happened as follows:

1. The Ochoco Forest Service livestock specialist, either by designation or by his own authority, announced the tree canopy cover has increased so there is less forage available for consumption. You will probably hear more about this. (For what it is worth those of us who actually own and manage livestock are not very impressed with his knowledge or some of his whacky decisions.)

2. A three-year process was initiated of public meetings where a new "management plan" was going to be formed. I want to comment on this a little. First, I understand this is a requirement from Washington. It is, however, simply to have a series of public meetings to get public input or a consensus. Your staff had interpreted it as do nothing for three years and at the end of that time the public meetings will tell them what to do and how to do it. Meanwhile the horse herd keeps expanding. The actual fact is your people have not a clue how to do all the technical stuff and a room full of horse lovers with almost no technical

expertise is going to get nothing technical done. What you are essentially saying is that you can have three years of public meetings with a bunch of housewives and at the end they will tell you how to build an atomic bomb! Meanwhile on your grazing allotments and in the local community you have many livestock experts, often with college degrees focused on ranching, who can figure out what needs to be done. Under pressure from several of us you started an inefficient and ineffective PZP treatment that may only slow the herd expansion by a single digit percentage. You needed to go all out for a permanent technical solution – YEARS AGO.

3. Stacy Forson arrived as the new Forest Supervisor. It appears as soon as she started to recognize there was a problem things started happening. At this time she has the full support of many of us. Frankly many of us feel she is the only hope of getting the problem fixed. We know the older staff well.

Next I want to cover the inbreeding issue. This may turn out to be the heart of your internal staff problem or the overall herd management problem. As I mentioned earlier the 55 to 65 AML will eventually suffer from severe inbreeding issues. (Actually the herd will eventually die off.) It also takes away from adoption attraction and general public viewing. Who wants to look at or adopt a freak? It appears that the COWHC and their agents on your local staff figured all this out some years ago. There was a problem and it was getting worse – probably rapidly. I actually started figuring this out after speaking with the more rabid COWHC members after the December 14 management plan meeting. They essentially indirectly gave their plan away. Here is what they probably knew:

1. You, as Regional Forrester, have the power to eliminate the Big Summit herd.

2. According to the rules (or Handbook) you are mandated to eliminate the herd if inbreeding cannot be controlled.

3. Both the National Wild Horse advocacy website(s) and the 2010 BLM Wild Horse Management Handbook have a formula for eliminating inbreeding problems. I will not look this up now but it is something like 150 to 200 breeding age females in the herd and adding two outside females every 10 years. I believe a breeding age female is between 5 and 10 years of age – but I am not sure.

In other words they desperately needed to increase the herd size to save the herd.

4. They could assume, probably correctly, that the Regional Forrester would not have the motivation or courage to kill horses. In other words if they got a big herd they could keep it. The horses are expected to live 16 to 18 years.

5. If they got a big herd the technology did not exist to reduce the numbers. They were aware of PZP but they were also aware you had no effective way of administering it 100%. They also knew PZP also has some shortcomings. (I will discuss this later in this letter.)

The COWHC has already successfully gotten your staff to add the two outside females.
This was evidently done several years ago.

Their next important goal was to do everything possible to increase the herd size. They have been incredibly successful in this endeavor. The three-year meeting delay was probably an integral part of their plan.

I have to give the COWHC credit for a very successful undercover, inside operation. Well done!

Several things have gone wrong with their plan as follows:

1. Your Ochoco staff stood by me on the sheep grazing. I have been getting reports and hearing rumors for a number of years that there were complaints by the horse people. (I assume it was COWHC members?) This would have been a major setback for they need the forage. (They may still get it but that will probably be up to you or Stacey.)

2. They misjudged the territory requirements of the horses. Forage shortages and harem issues will cause a wild horse herd to expand their territory. The harem issue is subtler. Stallions will fight hard to establish a harem – even to the death. Once one is established they will want to take it away from other stallions or the potential for fighting. Normally the harems will be small bands of 6 to 12 horses. This can cause herd area expansion - even if there is good forage.

3. They did not expect horses to start leaking onto private property. They probably assumed the horses would expand on Forest Service land. (They have figured this out and I mentioned the response of your COWHC influenced staff earlier.)

4. They did not expect Stacey and me. They now have a lot of other people getting involved including a few landowners. Until this happened they more or less had full control of the Ochoco Forest Service policy through their agents.

5. In 2013 the National Academy of Science did a book for the BLM that totally eclipsed the 2010 BLM Handbook and other wild horse advocates recommendations. I will discuss the book below. I gave a copy to Stacey. Its existence or the fact that it was in the Forest Service's hands was quickly leaked to the COWHC. They might have known about the study earlier? It totally blows up their plan. (They have countered and are trying to get your people to use a different "expert". At the December 14 meeting the COWHC already had their "expert" firmly planted with your staff. I shot most of this down with a December 29 letter. I exposed the COWHC "expert" as a fraud. The last I heard the COWHC objects to the "expert" I will recommend below who is the one the BLM uses and is widely quoted in the above book.)

You might be beginning to wonder how I am getting all my information. I will explain what I did. At the December 14 meeting I listened to the COWHC people speak and afterword had a conversation with them. I next spoke with some of your staff and quickly realized they were letting the COWHC run their show. ("Partners".) I then started a lot of Internet searches that eventually led to some Case Law using "Pacer". My first big breakthrough was picking up the phone and calling the BLM guy in Reno who writes *their National Newsletters. Most of the information or leads herein came from him in about a half hour conversation. At his recommendation I looked up the 2013 book, the appropriations bill, and some other stuff. I then, at his recommendation, visited the Burns Wild Horse facility and spoke with their experts. That was also very informative. The rest of the stuff was simply follow-up.*

Below is the 2013 book. You can actually save it from the Internet for free:

http://www.nap.edu/catalog/13511/using-science-to-improve-the-blm-wild-horseand-burro-program

Starting about page 141 is the inbreeding stuff. The expert the BLM uses is: E. Gus Cothran, Ph.D., Director Animal Genetics Laboratory of the Veterinary Integrative Biosciences Department of Texas A&M University. Dr. Cothran is probably the world's leading expert on Western United States Wild Horses. Again, the COWHC has already objected to him, as he might disrupt their plan. If you want to read the above book you can get more information – but I warn it is hard reading!

To use Dr. Cothran you are probably going to need to collect current samples from the majority of your horses. This will take a long time.

I believe it would be wise to contact Dr. Cothran as soon as possible and get a preliminary opinion.

I want to emphasize that I have no idea what Dr. Cothran's final recommendation for herd size would be. It could be a 50-horse herd or it could be a 600-horse herd. I believe it will all depend on the current inbreeding "number" or status.
If you decide on Dr. Cothran the whole project should be put under his control.

Later in this letter I will make another recommendation that totally bypasses Dr. Cothran and the processes designed for large wild horse herds on open desert type terrain. It totally fits your situation. It is based on conventional genetic management.

Next I want to spend a short time on the amount of money you are willing to spend on the horses and the public benefit factor.

By my calculation you will need to capture every horse you have at the very least every other year. This will be very difficult if not almost impossible. This will need to go on almost indefinitely as at this time you need strict birth control. Furthermore, this is a complete unknown because you also own a significant number of horses on private property and on places they are not supposed to be.

If you decide to officially increase your AML size this is adding a significant annual cost - forever.

If you go with the COWHC recommended horse area expansion this is going to be an additional cost, as you must move your traps over a larger area.

To give you a benchmark I understand you were paying between $700.00 and $1,000.00 per horse to have them randomly captured and trucked to Burns.

I also recently read that the BLM paid nearly one million dollars to have the infamous Bundy herd of around 900 cattle in Nevada rounded up. (Cattle are easy to gather – but evidently not for the Government.)

The BLM in Burns quoted somewhere around the $700.00 dollar figure per horse and they do their own work with very experienced people. This figure included trucking them to the Burns facility and processing them with very efficient equipment.
Since you need close to 100% capture rather than random capture the figures above might be much higher.

You also have other overhead costs related to the horses. From what little I know about these costs they could be reduced somewhat by some policy changes.
It would not be unreasonable to guess you will be in the six figures per year range.
Briefly Oregon Fish and Wildlife, hunters, livestock people, campers who do not appreciate all the horse manure on certain camping areas, adjoining private property owners impacted by the horses, and probably others do not like the herd expansion or the present numbers. These parties consider for the most part your horse herd feral and an invasive species.

Another way to look at it is would the general public like to see a horse or an elk or deer?

Lets look at the use.

You have an excellent web site advertising the wild horses:
https://fs.usda.gov/Internet/FSE_DOCUMENTS/fsbdev3_035796.pdf

Recently in a public meeting in Redmond a COWHC leader stated that people are coming from all over the world to see your horses.

From about June 17 through about September 18 I have two herded bands of sheep in the area and one camp tender that is driving in the area most days. In a way we might have more daily presence than the Forest Service Staff during that time period. This is not by any means an accurate survey but here is what we have seen: There is a small group of what appears to be the same women who are seen frequently year after year. Occasionally they have seen larger groups who are counting the horses. Other than that they see very little traffic and what traffic they see does not appear to be specifically looking for wild horses or necessarily stopping to observe when they do see them.
Others who are working nearby or are land owners report about the same thing.
I know years ago some Government Agencies had a method of putting a value in dollars on a daily visitor to a public facility. I suggest you might have a serious problem in this area? I do not think there is an awful lot of general interest in your horses. For sure there is not enough interest to support a six figure annual budget or expense!

The next subject I want to cover is capture, identification, and counting/record keeping. I am going to try to make this brief for the entire subject is quite complex.

Keep in mind you are going to need to approach 100% capture during a single year. This will be discussed in detail later in this letter.

The Burns Wild Horse Facility uses helicopter and trip-wire technology for capture. Their horses are very wild. They are for the most part in open desert. Their horses get wise to the helicopter method so it is not 100% effective. They have the same problem with their trip-wire technology. Some horses can never be caught with this method and once the horse becomes trap wise or has been trapped once it is no longer effective. Furthermore, due to distances their trip wire technology is very expensive to operate because traps must be checked daily.

You already have trip-wire technology for the Big Summit herd. It works and you are administering PZP shots with a dart rifle. It is also expensive to operate due to required daily checks and driving. Since you will need to do multiple captures making horses trap shy and the fact that some horses are even trap shy from the start you will not be able to come even close to 100% capture with your present equipment.

Incidentally in your letter you used the word "experiment" relating to the current use of your trip-wire technology and administrating PZP shots. Wrong. The technology was already proven. My criticism is that you have higher priorities for your limited resources. This is not really an important point. I am pleased that you are at least trying to do something.

Most of the horses the Burns BLM deals with are truly "wild". They have been loose since the Spanish first arrived in area. Their breed has an actual name and identified DNA. The Big Summit herd is recently abandoned domestic horses. Furthermore, there is a lot of human traffic in the area. My Herders report they can walk up close to some of them. They come into our camps at night looking for minerals. This will help trapping and processing.

The Big Summit terrain is a unique challenge. All the herding and counting methods used by the BLM or in other herds will not work due to the tall pine and fir trees or a dense forest setting.

You are probably going to be approached with several different capture and counting schemes. Their probability of working is almost zero. Be careful!
I strongly recommend you direct all your resources toward modifying the new, cloud based technology for remote control of traps that includes a camera system. The technology was initially developed for trapping the very intelligent, very trap shy wild pigs. It has been successfully modified for selectively trapping elk in Oregon.
This system is very portable. This is a requirement for your needs.

A cell phone or a computer operates the system remotely. A camera is operated by manual command, a timed schedule, or by a motion sensor. The camera also works at night. The trap is closed by a remote command – normally from a cell phone.
You can operate the system from Portland. You simply watch until the trap has the desired horses in and then, using your cell phone, shut the gates. Then you drive to the trap.

As trap shy horses watch other horses enter the trap safely for a number of days they will enter.

There is a lot more to this. Your Ochoco Staff has all the detailed information so the project is ready to go NOW.

The next critical issue is being able to identify an individual horse. This is for both the count for management and to be able to have an accurate record on each horse for age, DNA results, and the sterilization/abortion shot records. This probably lends itself to a computer database like Excel or Filemaker.

The Burns BLM facility uses necklace numbered tags inside their facility for identification. They do a neck, coded freeze brand on all horses sent out for adoption. This is so people will not adopt horses and return them to the wild. They also neuter all males they capture and put in their facility for adoption.

I would mention the Burns BLM has a very expensive processing facility and even very expensive portable processing equipment. You could spend 30K to 50K replicating their portable equipment. With their equipment you could mark horses with a small tattoo or even a small metal ear tag system. You can totally restrain the horse. For your application that includes multiple gathers and a trap shy issue. I doubt this system would be cost effective for such a small herd. Every time you captured a horse you would need to restrain it to get its number. As a last resort it is an option – but in my judgment a bad one especially since the horses could become very trap shy.

Using the system you have that has been modified for the new technology you will need to:

1. Administer all shots from outside the trap using a dart gun technology. Safety is not an issue.

2. Get inside the pen with horses once in the horse's lifetime. This is to mouth for age, take the required blood/DNA samples, and install a marking system. This needs to be done using tranquilizer technology. Safety is an issue.
At the present time your Ochoco Staff and the COWHC have come up with an identification system based on written records and photography. I do not know exactly how their system works, but I do know that it will not work. It is not accurate. Simply put if you continue along this path your herd expansion will continue as you waste a lot of time and resources. I can back this up with a lot of livestock experts.
I have suggested a small ear tag or freeze branding. Either identification system has to be readable in your capture pen while standing outside of the pen.

Freeze branding is fairly difficult and expensive. The horse will need to be almost completely put to sleep as the process is long and the horse must be held very still. It almost requires the expensive processing equipment the BLM uses.

On the other hand the small ear tag technology is inexpensive, permanent, and easy to insert in a moderately tranquilized horse.

Someone on your staff or possibly the COWHC told the District Ranger “Washington has said you cannot use ear tags”. Upon repeated requests he has not verified the exact source. Frustrated, I checked with the BLM and ear tags are allowed for them.

Furthermore, ear tags are widely used for horses other places. I suggest you use the small ear tag technology. This may need your approval?

Next I will briefly go over the art of the actual trapping technique.

Bait will be a big issue. Between late April and late August it will be difficult to bait because the natural forage is growing and very nutritious. In other words effective trapping may be seasonal. Unless there is a lot of snow the horses will be spread out and there will probably be a harem issue. Therefore the trap will probably need to be moved a lot to get close to 100% capture.

It is very unlikely the trapping will all be possible during a short time frame. It would be better to plan on it taking a long time. Furthermore, it appears the most efficient trapping method would be to do it every other year.

I trap a lot on my ranch for coyotes. They do not mix well with small lambs. We write everything down. I would strongly suggest you follow the same technique. It seems trapping is a series of small tricks that are often site specific.

Lets quickly sum up where you are at this point. You can capture horses inexpensively. You can get in the pen to do the sampling, marking, mouthing, etc. using tranquilizer technology. You can safely and quickly administer shots from outside the pen. You have records on each individual horse and an easy way to accurately identify each one. You can even follow the law and euthanize captured horses in the pen. Last you can probably come close to 100% annual capture using the remote controlled system outlined above.

Now we come to PZP. The Big Summit herd is a good candidate for PZP. Why? Almost 100% annual capture is possible. This means a slow decrease in the herd number is possible using PZP.

There are two options because PZP has only a 22-month life. They are:

1. A PZP shot each year.
2. A PZP shot every other year with an abortion shot added if the last shot was over 22-months past the current shot.

One of the above methods should be quickly started for the specific purpose of getting a good count and taking samples so a final inbreeding solution can be made. The herd reduction will be started. A few old or problem horses can even be euthanized as required by law.

From briefly reading the 2013 Book mentioned earlier there is a high probability that it will be recommended that to avoid inbreeding a greatly expanded horse herd will be required. The sheep allotments will need to go and possibly some cattle allotments. There will surely be an issue with private property owners. I believe you can push parts of this decision down onto Stacey, but I would suggest the ultimate decision would fall on you or someone higher up. Below are you options:

1. Turn the Ochoco National Forest into a horse farm. You will need a huge budget.

2. Continue with the herd as is or with a slow herd reduction using PZP and let the inbreeding happen. After a number of years or decades the herd will probably die off.

3. Follow the method below.
This method is conventional livestock genetics. It has been tried by the BLM but was not successful after a few years because they did not follow up with PZP or a second capture. The unborn males at the time of the neutering grew up and the problem returned.

Essentially you neuter all males born within the herd. You continue with the PZP or PZP/abortion shot method. Any active males or untagged males found in the herd must be quickly shot or neutered. After the desired numbers are reached carefully selected males from outside the herd are introduced for a specific time period and then neutered. Under no circumstance is any active stallion allowed to be with the herd over two years.

The neutering procedure is a 5-minute procedure that needs to be done under the supervision of a veterinarian. It is done on the ground in the regular corral. This is the method used by the BLM.

Here are the disadvantages:

1. The COWHC will object.

2. The “wildness” factor will in part be removed. For example stallions will not be fighting to the death but instead males will be docile and closer to 50% of the herd.

3. Management will be necessary. The current, inexpensive status of today will be gone.
4. Strict supervision will be required to make sure no active stallions are introduced to the herd or skipped during capture.

5. The viewing public will have to look harder to find horses.

Here are the advantages:

1. The inbreeding issue is gone.

2. The AML and allowable territory can remain as they were set in 1975.

3. With careful breeding the herd genetics can be excellent. If you want to save money this would be a great task for the COWHC. My suggestion would be to use Dr. Cothran as one of his studies would be the breed makeup of your herd.

4. The migration onto private property will probably stop or at least slow as there will be no forage and harem issues. This will take a few years.

5. Eventually, the annual cost will be very low and justifiable related to the anticipated general public use. To get to that point, however, could cost over a half million dollars.

6. The overwhelming majority of the stakeholders will be very pleased.
Ms. Li in my last letters I mentioned I was not represented by an attorney and was writing as a simple rancher. That might have been a lie. See the two cases below. Note the quality of my legal representation. In the first case I got mad – really mad. In the second case a poorly qualified, biased local judge screwed me over.

http://www.ktvz.com/news/coming-up-jury-awards-hefty-damages-in-ochocossheep-dog-killings/38450872
http://caselaw.findlaw.com/or-court-of-appeals/1490634.html

Both cases set a precedent. I think I might have done overkill in the first case?
Mr. Peña, if you carefully read this entire letter you probably have a headache. I hope I have driven home the complexity and depth of the situation you are in. Your local Ochoco staff needs outside help.

If it helps just keep in mind there are several livestock trucks leaving Prineville empty each week bound for Canada to pick up pigs to haul to Klamath Falls for slaughter.

According to the very clear current law these trucks can take all or parts of your problems with them!

Thanks,
[redacted]
cc: Stacey Forson

Call with Dr. Gus Gothran

September 13, 2017

Prepared by Steve Gibson

We began by discussing genetic diversity of the Big Summit herd based upon Dr. Cothran's 2011 report and Dr. Mill's 2011 manuscript and that the Ochoco had introduced 2 mares from the Steens HMA based upon the findings of both of these papers. Dr. Cothran indicated that allelic variability is highly dependent upon sample size, and since there was a small sample size in both papers one would expect small variability as a result. However, observed heterozygosity is independent of sample size and this measure was also quite low for the Big Summit herd samples indicating probable inbreeding.

When asked about scoping comments attributing a 150 head minimum viable wild horse population to him, he responded with an explanation of the origin of the determination. He was involved in the advent of conservation genetics primarily associated with endangered species. At the time (I believe it was early 80s), experts in this particular area reached consensus that an acceptable rate of loss of genetic variability to maintain a species for 2000 generations was 1% per year. This equates to 50 effective breeding individuals. The number of effective breeding individuals was considered a third to a quarter of a local wild horse population. Thus the number of 150 to 200 head was arrived at. When he was working with determining a minimum viable population level for the Pryor Mountain herd they used the 150 head number because genetic sampling indicated the herds genetics represent a great deal of variability. He did point out that if an isolated group of horses is already lacking genetic variability that simply allowing that population to increase in number will not increase variability but will continue to experience loss of genetic variability and inbreeding depression.

While discussing evidence in these reports of ties to Old World Iberian ancestry and genetic uniqueness requiring special treatment, Dr. Cothran made the following observations. While the small sample size did appear to cluster genetic similarity around the Old World Iberian breeds, mean similarities to New World Iberian breeds and Oriental and Arabian breeds were within 0.005 of Old World Iberian breeds, a negligible difference when the standard deviation is between 0.2 and 0.3. In addition, Dr. Cothran pointed out that 0.62 is a low value for similarity indicating that there is not a definitive ancestral relationship established based upon the limited sample size represented. While clustered around the Andalusian breed, Dr. Cothran indicated that the ladder chart represented in Figure 1 of his paper, placing these horses between pony breeds, is not satisfactory to him. I shared, the anecdotal story of someone dumping a trailer full of Shetland ponies onto the territory in the distant past and that just based on physical appearance the Big Summit horses appear to be an interesting mix of pony and draft horse. Dr. Cothran responded that in other herd(s) he has worked with in the past he has found that herd(s) with a pony, draft and mid-size American horse breed (like quarter horse) the similarity index displayed Andalusian association under very small sample size but resolved with more samples.

Dr. Cothran reiterated that what we are confident of based on these samples is that genetic variability is limited. Dr. Cothran indicated that if Dr. Mills was willing to share her samples his laboratory could run the results rapidly (few days) which would increase the overall sample size. He indicated if she wanted to publish anything based upon the results from his lab he would be fine with it. Cost would be in the neighborhood of 40-50 dollars per sample.

Dr. Cothran stated he will often take issue with use of the term “unique” in describing the genetics of a locally isolated population of organisms as each will have a specific suite of ancestral and environmental selective pressures exerted upon it that is not replicated elsewhere. He indicated that in making a determination as to whether the Big Summit herd is unique in a way that requires special treatment in management of the herd he would have to respond with a “no”. As there is not enough information available to make that determination. Regardless, the agency would be very hard pressed to come up with a way to maintain genetic variability in the herd without periodic introductions of individuals from outside herd management areas. His recommendation for managing the genetics of our herd based upon existing information is to maintain the herd at the maximum level provided by the resources in the territory, and introduce a new mare to the population every two to four years.

When asked about the National Research Council’s statement in the genetics chapter of Using Science to Improve the BLM Wild Horse and Burro Program, “… empirical work suggests that if maintenance of fitness is important, effective population sizes much larger than 50 are necessary. Theoretical studies suggest that the figure could be closer to 5,000 …”, Dr. Cothran indicated that every conservation geneticist will state that more is always better and that we need to maintain as many as we can, however, environmental factors will often limit populations long before they reach 5,000 effective breeding individuals.

Cothran, E. Gus. 2011. Genetic analysis of the Big Summit HMA, OR. Department of Veterinary Integrative Bioscience, Texas A&M University, College Station, Texas.

Mills, Dr. DeEtta. 2010. 2010 Report to the Prineville USDA-Forest Service Office, Prineville, OR: The genetic analysis of the Ochoco National Forest wild horses. Florida International University, Miami, Florida.

National Research Council. 2013. Using Science to Improve the Wild Horse and Burro Program: A Way Forward. Washington, DC: The National Academies Press. https://doi.org/10.17226/13511.

OCHOCO WILD HORSE HERD MANAGEMENT PLAN OBJECTIONS

Melinda Kestler

Appendix B

Strategic Research Plan Wild Horse and Burro Management, 2003 revised 2005, The BLM Wild Horse and Burro Program U.S. Department of Interior, Pgs. 1-45, PDF.

The 2020 Five Domains Model: Including Human–Animal Interactions in Assessments of Animal Welfare, Pgs. 1-24, PDF.

Seasonal Variation in Habitat Selection by Free-Ranging Feral Horses Within Alberta's Forest Reserve, Pgs. 428–437, PDF.

Management Implications of the Ecology of free-roaming horses in Semi-arid Ecosystems of the western United States, Erik Beever Pgs. 887-895, PDF.

Background for NEPA Reviewers: Grazing on Federal Lands, U.S. Environmental Protection Agency, Pg. 1-39, PDF.

animals

Review

The 2020 Five Domains Model: Including Human–Animal Interactions in Assessments of Animal Welfare

David J. Mellor [1,*], Ngaio J. Beausoleil [1], Katherine E. Littlewood [1], Andrew N. McLean [2], Paul D. McGreevy [3], Bidda Jones [3,4] and Cristina Wilkins [5]

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Simple Summary: This review outlines the latest in a succession of updates of the Five Domains Model, which, at each stage, incorporated contemporary verified scientific thinking of relevance to animal welfare assessment. The current update includes, within the structure of the Model, specific guidance on how to evaluate the negative and/or positive impacts of human behaviour on animal welfare. Persons whose actions may be evaluated include, but are not limited to, livestock handlers, owners of draught animals, veterinary care staff, pound/shelter staff, zoo-keepers, wildlife managers, hunters, researchers, companion animal owners, owners of sport/recreational animals, animal trainers and service animal handlers. Situations where human–animal interactions may have negative welfare impacts include: when animals have had little or no prior human contact, when human presence adds to already threatening circumstances, when human actions are directly unpleasant, threatening and/or noxious, when humans' prior actions are remembered as being aversive or noxious and when the actions of bonded humans cause unintended harms. In contrast, situations where human–animal interactions may have positive welfare impacts include: when the companionable presence of humans provides company and feelings of safety, when humans provide preferred foods, tactile contacts and/or training reinforcements, when humans participate in enjoyable routine activities or in engaging variable activities, when the presence of familiar humans is calming in threatening circumstances and when humans act to end periods of deprivation, inhibition or harm. The explicit delineation within the Model of the potential impacts of human interactions on the welfare of animals enhances the Model's utility. Additional updates in this latest version are also explained.

Abstract: Throughout its 25-year history, the Five Domains Model for animal welfare assessment has been regularly updated to include at each stage the latest authenticated developments in animal welfare science thinking. The domains of the most up-to-date Model described here are: 1 Nutrition, 2 Physical Environment, 3 Health, 4 Behavioural Interactions and 5 Mental State. The first four domains focus attention on factors that give rise to specific negative or positive subjective experiences (affects), which contribute to the animal's mental state, as evaluated in Domain 5. More specifically, the first three domains focus mainly on factors that disturb or disrupt particular features of the body's internal stability. Each disturbed or disrupted feature generates sensory inputs which are processed by the brain to form specific negative affects, and these affects are associated with behaviours that act to restore the body's internal stability. As each such behaviour is essential for the survival of the animal, the affects associated with them are collectively referred to as "survival-critical affects". In contrast, Domain 4, now named Behavioural Interactions, focusses on evidence of animals consciously seeking specific goals when interacting behaviourally with (1) the environment, (2) other non-human animals and (3) as a new feature of the Model outlined here, humans. The associated affects, evaluated via Domain 5, are mainly generated by brain processing of sensory inputs elicited by external stimuli. The success of the animals' behavioural attempts to achieve their chosen goals is reflected in whether the associated affects are negative or positive. Collectively referred to as "situation-related affects", these outcomes are understood to contribute to animals' perceptions of their external circumstances. These observations reveal a key distinction between the way survival-critical and situation-related affects influence animals' aligned behaviours. The former mainly reflect compelling motivations to engage in genetically embedded behavioural responses, whereas the latter mainly involve conscious behavioural choices which are the hallmarks of agency. Finally, numerous examples of human–animal interactions and their attendant affects are described, and the qualitative grading of interactions that generate negative or positive affect is also illustrated.

Keywords: affective state; biological functioning; behavioural interactions; human behaviour; environment; other animals; humans; welfare impacts; welfare grading

1. Introduction

The Five Domains Model for animal welfare assessment was originally formulated in 1994 [1]. It was subsequently updated in 2001 [2], 2004 [3], 2009 [4], 2012 [5], 2015 [6] and 2017 [7] to incorporate current, authenticated developments in animal welfare science thinking. The associated evolution of the Model is outlined in detail in Section 2. In general terms, the updates incorporated contemporary knowledge of interactions between physiological mechanisms and the generation of particular subjective experiences, known as affects or affective states. They also expanded the range of specific affects to be considered and clarified their biological roles. Initially, the emphasis was on welfare-compromising negative affects, and later, welfare-enhancing positive affects. Finally, the methodology for undertaking Model-based welfare assessments was refined as the Model was increasingly applied internationally to wider ranges of vertebrate species and animal use sectors.

The aim of this review is to include, within the structure of the Model, specific guidance on how to evaluate the negative and/or positive welfare impacts of human proximity to and/or behaviour towards animals. Although all published versions of the Model have included brief reference to such human impacts, usually they were portrayed simply as being potentially aversive, neutral or benign. However, during the last 5–10 years, increasing attention has been given to conducting much more detailed assessments of such impacts. The persons of interest include livestock handlers, owners of draught animals, veterinary care staff, pound/shelter staff, zoo-keepers, wildlife managers, hunters, researchers, companion animal owners, owners of sport/recreational animals, animal trainers and service animal handlers. Accordingly, the Model has been extended to facilitate explicit and detailed assessment of the welfare impacts that these people may have on the animals in their care or control.

The current review begins, as indicated above, with an account of the principal features of the ~25-year evolution of the Model (Section 2). The general features of the 2015 Model and the methodologies for grading welfare compromise and enhancement are then described (Section 3). The rest of the review focusses on the 2020 Model. Details of the first three domains (1 Nutrition, 2 Physical Environment and 3 Health), are outlined, updated and the features they have in common are identified (Section 4). We also show how a range of factors in each domain generate specific negative or positive affects that are evaluated via Domain 5, the animal's Mental State. Domain 4 and its attendant Domain 5 affects are then described in detail (Section 5). Previously called "Behaviour" and now

"Behavioural Interactions", Domain 4 is subdivided according to the nature of animals' interactions with (1) their environment, (2) other non-human animals and (3) humans. The last of these is described extensively, including consideration of the grading of negative and positive welfare impacts. Finally, the review ends with concluding comments (Section 6).

2. The 25-Year History of the Five Domains Model: Responses to Changes in Animal Welfare Thinking

2.1. Formulation of the Model for Assessing Negative Impacts of Research, Teaching and Testing

The Model, originally formulated in 1994, had the specific purpose of prospectively and retrospectively assessing and grading the negative impacts of research, teaching and testing (RTT) procedures on sentient animals [1]. Deployment of the Model enabled such assessments to be made in much greater detail than before [1,3,8–10]. In 1997, assessments and grading using the Model were mandated within the regulations that govern animal ethics committee scrutiny of all proposed and completed RTT activities in New Zealand [10], a requirement that continues to this day.

Prior to formulation of the Model, RTT impact assessments usually focussed very narrowly on the precise details of the particular manipulation(s) to be applied to the animals, leaving largely unexamined the animals' wider circumstances that could cause additional negative impacts [1,8–10]. The first four of the five domains of the Model were developed to correct this [1]. Their purpose was to draw attention both to interactions among diverse functions within the body and to the negative impact of external factors on those functions, all of which, in various combinations, have relevance to impact assessments across the four physical/functional domains. Finally, the fifth domain was designed to capture the overall mental experience of the animals, evaluated in terms of the suffering from all impacts considered within the first four domains [1]. Hence, the explicit focus of the 1994 Model was the detailed and holistic assessment of animal welfare *compromise*. It also provided a basis for qualitatively grading the severity of the negative impacts [1,3].

The five domains were: (1) nutrition, (2) environment, (3) health, (4) behaviour and (5) mental state. The first three domains focussed attention on *internal* imbalances or disturbances which had nutritional, environmental and health origins. In contrast, the focus of the fourth domain was on *external* restrictive confinement or restraint, or otherwise unusual space availability and/or negative impacts of the presence or absence of other animals (including humans) [1,3]. After collation of the objective evidence derived from consideration of factors in the first four domains, the subjective, emotional or affective experiences, cautiously inferred to be associated with these disturbances or restrictions, were then assigned to the fifth domain [1,3,4]. The fifth domain enables an ultimate assessment of the overall welfare state of the animals, understood in terms of what they were likely to experience subjectively (Figure 1). Notably, the first version of the Model restricted these experiences to thirst, hunger, anxiety, fear, pain and, as a catchall term, 'distress' (Figure 1) [1].

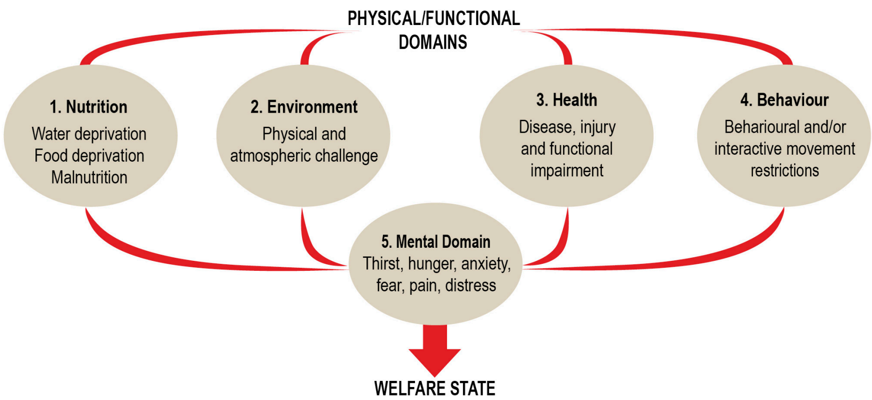

Figure 1. The 1994 Five Domains Model, redrawn from Reference [1].

From the outset, the Model was based on the premise that physiological mechanisms, later generalised in the term "biological functioning" [11], are the foundation of affective experiences, that affective experiences can influence physiological mechanisms and that both of these elements interact dynamically within the body which operates as an integrated whole entity [1]. However, around this time, two competing schools of thought emerged, one emphasising "biological functioning" and the other "affective state", each of them arguing that the other had significant shortcomings in the ways it assessed animal welfare (see References [7,11,12]). Now, it is widely recognised that these two elements interact dynamically and that together they provide a more comprehensive foundation upon which to base welfare assessments [7,13–17]. Also, the inclusion of Domain 5, mental state, within the Model emphasises that what matters to animals in welfare terms is their subjective experiences. The 1994 Model therefore had an affective state orientation, but with the advantage, carried forward into all later versions of the Model, that the dynamically integrated alignment of the physiological mechanisms underlying specific affects provided a more coherent and informative basis for evaluating their welfare significance.

2.2. The Initial Emphasis on Negative Welfare States

The inception of animal welfare science occurred when the scientific method was first applied to evaluating problems perceived to have welfare significance [13]; for example, those in production animals exposed to inadequate nutrition, shelter/shade/space and protection against disease and injury [4,18–20]. From the outset, animal welfare science focussed on the optimal care of animals' physical/functional states, the aim being to be free of any identified problems [4,12,16,18,19,21]. In animal welfare terms, this meant that, for about 15 years, virtually all scientific attention was focussed on studying negative welfare states and the circumstances that caused animals to have unpleasant or aversive experiences [4]. The impacts of this approach were profound. It resulted in major science-based advances in understanding of animal welfare and its management (e.g., References [4,13,18–20,22–29]), advances which provided the foundations for the subsequent developments in thinking and ways of assessing welfare states, some of which are described below.

2.3. Giving Greater Definition to the Meaning of "Distress"

As noted above, the fifth domain of the 1994 Model drew attention to a limited range of specific affective experiences and, as a catchall term, to "distress" (Figure 1) [1]. This was deliberate because the inclusion of additional specific affects was thought likely to hinder acceptance of the Model at a time when the legitimacy of focusing on affective states had not yet been widely accepted among animal welfare and other animal-based scientists [11,13,14]. So "distress" became a "place-holder" for other negative affects animals may experience.

Use of this term, and equally generic references to "suffering", are still common today in animal welfare legislation, codes of welfare and legally enforceable regulations, and are also included in industry and institutional guidelines [30,31]. Nevertheless, it was increasingly recognised that such generic terminology can oversimplify the way animal welfare is formally and informally evaluated and regulated (see below). Accordingly, over at least the last 20 years, considerable attention has been given to identifying specific affects that may be included in these terms [6,16,25,31–34]. Thus, the list expanded, and in addition, two major categories of negative affect were identified [6].

The first category, survival-critical negative affects, refers to experiences generated by sensory inputs that register imbalances or disruptions in the *internal* physical/functional state of animals. They include breathlessness, thirst, hunger, pain (~30 varieties), nausea, dizziness, debility, weakness and sickness [4,6,23,32,35–39]. These affects are designated as survival-critical because they are aligned with essential components of genetically embedded mechanisms that elicit or are associated with behaviours on which the survival of the animals depends [35,37,38].

The undoubted negativity of each affect creates a sense of urgency, or a dominating compulsion, to engage in behaviours which are specific to that affect and its resolution (e.g., References [35,38]).

Examples of links between affects and responses include breathlessness and respiratory activity, thirst and water seeking/drinking, hunger and food acquisition, pain and escape or avoidance responses to injury, as well as weakness/sickness and securing benefits from isolation and rest [4,6,23,32,36,38,40,41]. Importantly, the greater the intensity of the negative affect, the greater the sense of urgency or compulsion to engage in the aligned behaviour, and vice versa. Once the behaviour achieves the required corrective physical/functional outcome, the intensity of the negative affect declines and, correspondingly, the motivation to perform the salient behaviour subsides [35,38]. Unpleasant experiences that cannot be effectively relieved through behavioural and physiological responses may have a greater detrimental impact on the welfare state than acute but short-lived experiences.

The second category, situation-related negative affects, refers to experiences generated by brain processing of sensory inputs that mainly originate from outside the body and reflect the animal's perception of its *external* circumstances, i.e., its situation [16,38]. These affects currently include frustration, anger, helplessness, loneliness, boredom, depression, anxiety, fear, panic and hypervigilance (see References [7,16,25,37,42–53]). Also note that the emotional pain of social isolation, i.e., loneliness, is now receiving increasing attention [52,54]. Animals in impoverished and/or threatening situations may experience these affects in various combinations.

The distinguishing attributes of each negative affect in these two categories have now been described [55]. Identifying the specific conditions that generate this wide range of negative affects and understanding the bases of their two categories, allows potential negative welfare impacts to be assessed more thoroughly and remedial actions to be focussed more precisely than before [6,16,31]. It is worth noting that the two categories are not mutually exclusive. For example, a tired racehorse that is being whipped may feel pain triggering escape or avoidance responses, and helplessness if those responses do not resolve the situation because the horse cannot escape the jockey who is the source of the pain [47,48]. Likewise, the experience of pain may be modulated by awareness of fear-inducing stimuli such as the presence of predators [56].

2.4. Including Consideration of Positive Affective Experiences in the Model

From the early 2000s, animal welfare scientists gave increasing attention to positive affective experiences (for References see [5,13,14,16,21,27,35,37,39,45,51,54,57–63]). This was motivated by the recognition that good or acceptable animal welfare, embodied in notions such as "a life worth living" [14,16,21,57], cannot be achieved simply by mitigating or avoiding negative experiences and that some pleasurable experiences are needed as well. Thus, attention increasingly shifted away from the mere care of animals towards their psychological well-being. The Model was therefore revised extensively to include, in each of the first four domains, the internal and external circumstances that may give rise to positive affects which, as with the negative affects referred to above, were assigned to the fifth (mental) domain for consideration [6]. Such experiences, when present, contribute to welfare enhancement.

These revisions were based on the scientifically supported understanding that animals may have pleasurable experiences when their external circumstances include, but are not limited to, the following: variability that provides an optimal balance between predictability/controllability and novelty/unpredictability, meeting species-specific needs for movement and exercise, access to preferred sites for resting, thermal comfort and elimination behaviours, environmental choices that encourage exploratory and foraging behaviours and durations, availability of a variety of feeds having attractive smells, tastes and textures, and circumstances that enable social species to engage as fully as possible in bonding activities with familiar conspecifics, the calming comfort of being in a group of familiar conspecifics and, as appropriate, other affiliative interactions such as allogrooming, bonding, maternal, paternal or group care of young, play behaviour and sexual activity [6,7,16,21,51,54,59,60] (see Section 5). Expressed in general terms, the associated welfare enhancing affects likely include various forms of comfort, pleasure, interest, attachment, confidence and a sense of being in control (see Section 5.2 on agency) [7,13,16,21,27,58,63,64].

2.5. Applying the Model to Numerous Species of Sentient Animals Evaluated for Diverse Purposes

Within the mandatory New Zealand regulatory context, the Model in its various updated versions (e.g., References [2–7,39]) has been used to assess the negative welfare impacts of RTT procedures applied to a wide range of sentient animals being evaluated for diverse purposes. As noted previously [12], the animals have included horses, cattle, deer, goats, sheep, pigs, poultry, game birds, other birds including endemic, native and introduced species, dogs, cats, guinea-pigs, mice, rats, rabbits, ferrets, stoats, weasels, kangaroos, wallabies, possums, cetaceans, reptiles, amphibians and fish. The studies' purposes have included fundamental and applied biomedical, veterinary, agricultural, ecological, welfare, educational and other approved investigations [12].

2.6. Expanding Application of the Model Beyond the Research, Teaching and Testing Context

In addition to these RTT purposes, the Model has also been used to prospectively and/or retrospectively assess negative and/or positive welfare impacts of proposed new or modified approaches to housing, managing and/or interacting with farm [4], working [65], livestock guarding [66], sport [67–69], zoo [4,70–74], wild [75], free roaming [76], introduced [56,77–82] and other terrestrial animals [4], as well as cetaceans [83,84]. The Model has also been used forensically in Canadian court cases to assess suffering and animal cruelty [55].

Given the diversity of animals and Model applications, there is merit in assembling scientifically informed experts who collectively can provide detailed input on species-specific biology, ethology, ecology, physiology, pathophysiology, health and management (e.g., Reference [85]), and also, affect-related, neuroscience-supported behavioural expertise, and experience with the operation of the Model [6,7,16,32,76,81]. Using widely experienced panels or consultative networks is helpful in such evaluations (e.g., References [58,66,67,70,78,80,82,84,86]).

3. The 2015 Five Domains Model

Full descriptions of the 2015 Model, including details of how it operates and its key applications to the assessment and management of animal welfare, have been published elsewhere [6,7,16,55,65]. It is strongly recommended that readers consult these sources after perusing the brief outline provided below.

3.1. General Features of the Model

The Model is not intended to define good and bad welfare, nor is it intended to accurately depict body structure and function. Rather, it is a device for facilitating systematic, structured, thorough and coherent assessments of animal welfare, and for qualitatively grading welfare compromise and enhancement (see Section 3.3) [6,7,16]. The purpose of each domain is to draw attention to areas that are relevant to welfare assessments, taking into consideration the understanding of animal welfare briefly outlined above and presented in more detail elsewhere [6,7,16,55,65].

In view of the dynamic interactivity of virtually all mechanisms in the body [7,14–17], there is inevitably considerable interaction among the specific body functions or states, the impacts of external circumstances and the related affective experiences identified via the Model. Accordingly, factors considered within different domains may overlap; for example, a painful event may be identified in Domains 2 and 3. However, when conducting a Model-based welfare assessment, the particular origin of a specific affect needs to be considered only once, so that it should be arbitrarily assigned to a single domain. This avoids concerns about duplication that may lead to over-weighting of a particular experience in the final interpretation, and it also avoids fruitless arguments about domain specificity.

The 2015 Model [6], in common with the 1994, 2001, 2004 and 2009 versions [1–4], is generic rather than species-specific. The primary purpose of the domains is to provide examples of some internal states or external circumstances that animals may encounter and the aligned negative and positive affects that may arise in many species. However, as particular affects generated by sensory modalities that are beyond direct human experience are not known, the details provided are neither definitive nor exhaustive. Examples include unique modalities such as echolocation, ultrasonic communication, infrared sensory abilities, electromagnetic field detection, highly adapted chemical and vibrational sensitivity, as well as the exaggerated or diminished acuity of the common modalities of vision, audition and olfaction across different taxa (for References see [87]), and also, the affective experience of flight in birds, bats and gliders. Moreover, essential information about some affects and their generation is very limited or non-existent in less well-studied animals, such as in many zoo or free-living wildlife species (e.g., References [70,72,84]). For example, it is not clear whether cartilaginous fish experience some kinds of pain because of the failure, as yet, to identify the necessary sensory receptors [88]. Accordingly, each example should be assessed by reference to what is known about the animals' species-specific behaviour, physiology and ecology considered in relation to its particular physical, biological and social environment [85].

The summary diagrams of the 2015 Model [6,7], all features of which have been included in the updated 2020 versions presented in Sections 4 and 5 (see Figures 2–5), have the status of guiding aides-mémoire. Therefore, when applying the Model to new species or contexts, the examples provided should be considered carefully and, only after sufficient justification, be retained, deleted or amended, and/or others added as deemed appropriate for each species (e.g., References [65,67,69,76–78,80,84]).

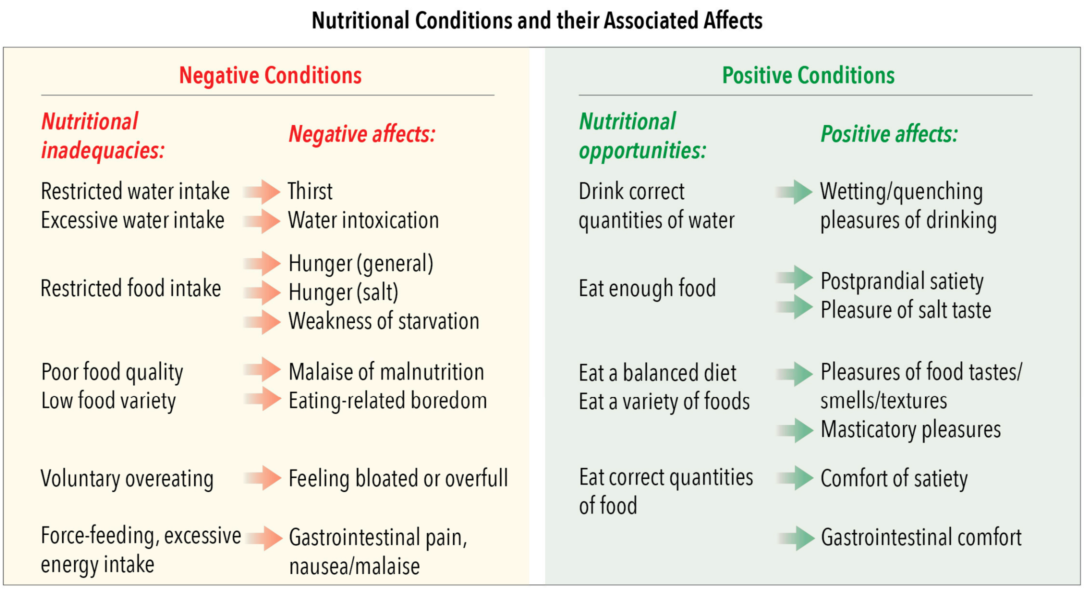

Figure 2. Domain 1: Nutrition. Examples of nutritional imbalances and opportunities and their associated negative and positive affects assigned to Domain 5: Mental State.

Finally, inclusion of environmental events or conditions in Domains 1 to 4 that *may* cause internally or externally derived imbalances or disruptions to the animals represent areas of increased *risk*, in which particular negative affects and welfare problems *may* arise [72,76]. However, their mere existence does not necessarily mean that the anticipated welfare problems will arise or have arisen in the particular situation under investigation. For example, the presence of potentially damaging structures in an animal's environment presents a risk of tissue injury but does not indicate that the animal is currently experiencing welfare compromise due to pain. Any assumption of the occurrence of negative affects must be supported by directly observed animal-based physical, physiological, clinical and/or behavioural evidence [39,76]. This is equally the case for the presence of *opportunities* for animals to engage in rewarding behaviours. Clearly, there must be evidence, usually behavioural, that any such opportunities are actually *used* before their potential welfare enhancing impacts could be considered. Only then can inferences be made about any aligned negative or positive affects. This emphasises the general point that objective animal-based evidence (Domains 1 to 4) must form the foundations of any inferences about welfare-relevant affects (Domain 5) [6,7].

Physical Environmental Conditions and their Associated Affects

Negative Conditions		Positive Conditions	
Unavoidable physical conditions:	*Negative affects - forms of discomfort:*	*Enhanced physical conditions:*	*Positive affects - forms of comfort:*
Close confinement; overcrowding	*Physical:* general stiffness, muscle tension	Space for spontaneous locomotion	*Physical* comfort
Unsuitable substrate, wet/soiled ground	*Physical:* musculoskeletal pain, skin irritation	Suitable substrate, well-drained ground	*Physical* comfort
Air pollutants: NH_3, CO_2, dust, smoke	*Respiratory:* breathlessness, air passage irritation/pain	Fresh air dissipates contaminants	*Respiratory* comfort
Aversive odours	*Olfactory:* revulsion at foul or repellent odours	Foul smells dissipated by fresh air & good hygiene	*Olfactory* comfort
Thermal extremes	*Thermal:* chilling, dampness, overheating	Effective shelter and shade available	*Thermal* comfort
Loud or otherwise unpleasant noise	*Auditory:* impaired hearing or ear pain	Effective noise control measures are in place	*Auditory* comfort
Light: inappropriate intensity	*Visual:* eye strain due to flashing, glare or darkness	Light intensity kept at tolerable levels	*Visual* comfort
Monotony: ambient, physical, lighting	Malaise from unnatural constancy	Within-day environmental variability maintained	Congenial variety and predictability
Unpredictable events	Anxiety, fear, hypervigilance	Predictability achieved by established routines	Relaxation-based ease and calmness
Physical limits on rest and sleep	Exhaustion	Conditions conducive to rest and sleep	Well rested

Figure 3. Domain 2: Physical Environment. Examples of unavoidable and enhanced physical conditions and their associated negative and positive affects assigned to Domain 5: Mental State.

Health Conditions and their Associated Affects

Negative Conditions		Positive Conditions	
Presence of:	*Negative affects:*	*Minimal or no:*	*Positive affects:*
Injury: acute, chronic, husbandry mutilations	Pain (many types), breathlessness, debility, weakness, sickness, malaise, nausea, dizziness	Injury	*Comfort* of good health and functional capacity
Disease: acute, chronic		Disease	*Comfort* of good health and functional capacity
Functional impairment: due to limb amputation, other therapies; genetic, lung, heart, vascular, kidney, gut, neural, or other problems		Functional impairment	*Comfort* of good health and functional capacity
Obesity or leanness: physical and metabolic consequences	Affects of being too fat or thin, and of metabolic and pathophysiological sequelae	Extreme body condition scores	*Comfort* of good health and functional capacity
Poisons	Many affects due to mode of action	Poisoning	*Comfort* of good health and functional capacity
Poor physical fitness, muscle de-conditioning	Physical weakness and exhaustion	Poor fitness (fitness level good)	Vitality of fitness and pleasurably vigorous exercise

Figure 4. Domain 3: Health. Examples of negative and positive health conditions and their corresponding affects assigned to Domain 5: Mental State.

Behavioural Interactions and their Associated Affects

INTERACTIONS WITH THE ENVIRONMENT			
Exercise of 'agency' is impeded:	***Negative affects:***	***Exercise of 'agency' is promoted***	***Positive affects:***
Invariant, barren, confined environment (ambient, physical, biotic)	Boredom, helplessness Depression, withdrawal	Varied, novel environment	Interested, pleasantly occupied
Inescapable sensory impositions	Various combinations: startled by unexpected events, neophobia, hypervigilance, anger, frustration, negative cognitive bias	Congenial sensory inputs	Likes novelty, post-inhibitory rebound
Choices markedly restricted		Available engaging choices	Calm, in control
Environment-focussed activity constrained		Free movement	Engaged by activity
Foraging drive impeded		Exploration, foraging	Energised, focussed
INTERACTIONS WITH OTHER ANIMALS			
Animal-to-animal interactive activity constrained	Loneliness, depression Yearning for company	Bonding/reaffirming bonds	Affectionate sociability
		Rearing young	Maternal, paternal or group rewards
	Thwarted desire to play	Playing	Excitation/playfulness
	Sexual frustration	Sexual activity	Sexually gratified
	Thwarted hunting drive	Hunting	Alert engagement, highly stimulated
Significant threats Limits on threat avoidance, escape or defensive activity	Anger, anxiety, fear, panic, insecurity, neophobia	Absence of threats Using refuges, retreat or defensive attack	Secure, protected, confident
Limitations on sleep/rest	Exhaustion	Sleep/rest sufficient	Energised, refreshed; post-inhibitory rebound
INTERACTIONS WITH HUMANS			
Negative human attributes and behaviour:	***Animal behaviours and negative affects:***	***Positive human attributes and behaviour:***	***Animal behaviours and positive affects:***
Attitude: uncertain, fearful, indifferent, insensitive, impatient, oppressive, belligerent, domineering, callous, cruel, vindictive *Voice:* hesitant, angry, loud, shouting *Aptitude:* inexperienced, unskilled, untrained, unqualified	*Behaviours (e.g.):* long flight distance, hypervigilant, attack/fight, hyper-reactive, escape avoidance, freezing, cowering, appeasing, withdrawn, non-compliant	*Attitude:* confident, caring, sensitive, patient, kind, empathetic *Voice:* confident, calm, clear, encouraging, pleasantly rhythmic *Aptitude:* experienced, skilled, trained, qualified	*Behaviours:* short flight distance, calm alertness, at ease with imposed hands-off or hands-on contact, compliantly responsive, explores novel events, seeks contact, variably bonded with humans
Handling/controlling: erratic, rough (slap, hit, kick, grab, poke, beat, whip); excessively forceful, violent; punishment-focussed; more negative pressure than is needed for training objective	*Affects:* anxiety, fear, panic, terror, neophobia; insecurity, confusion, uncertainty, persistent unease; helplessness; pain from injuries; negative cognitive bias	*Handling/controlling:* skillful, gentle (stroke, touch, push, guide); firm, temperate, restrained; reward-focussed; mimics allo-grooming by conspecifics; using subtle pressure cues, secondary reinforcers and timely release of aversive stimuli	*Affects:* calm, confident, at ease, feels in control; enjoys variety; finds being bonded with humans rewarding

Figure 5. Domain 4: Behavioural Interactions. Examples of interactions with the environment, other (non-human) animals and humans, where animals' capability to freely exercise agency would be impeded or enhanced, and examples of the corresponding affects assigned to Domain 5: Mental State. Also provided for human–animal interactions are examples of negative and positive attributes which influence the behaviour of humans towards animals.

3.2. Summary of the Grading Methodology of the 2015 Model

Grading systems have been incorporated into the Model from its original formulation [1,4,6,10,65]. The bases for grading negative and positive welfare impacts differ. The defining point of reference for welfare compromise is suffering and its mitigation, whereas the focus for welfare enhancement is on animals' use of opportunities to experience positive affective engagement [6,7,59]. The corresponding welfare impact scales also differ.

A five-tier scale (A to E) is used to grade negative welfare impacts according to the presence, intensity and/or duration of specific negative affects. Thus, grades A and B represent no and tolerably low-level impacts respectively, grade E represents very severe negative impacts related to experienced affects variously manifesting at high to very-high intensities and/or for long to very-long durations and grades C and D represent intermediate-level impacts related to their intensities and/or durations. These grades therefore equate to different degrees of welfare compromise, ranging from none to very severe [4].

Although a five-tier scale is notionally available, this does not mean that in all cases grading can be achieved with the degree of precision implied by that number of tiers. For example, when information is limited or contradictory, it may be possible to distinguish only between no to low, moderate and severe negative impacts, or, at its simplest, when a particular impact is either absent or present [6,80–82,89]. From the outset, numerical grading was explicitly rejected to emphasise the importance of using scientifically informed judgement, and to avoid implying, unrealistically, that much greater precision is achievable than is actually possible with such qualitative assessments [1,7,10].

In contrast, a four-tier scale (0, +, ++, +++) modified from that developed by Edgar and colleagues for poultry [58], is used to grade positive impacts where the tiers represent no, low-level, medium-level and high-level enhancement, respectively. This scale has three integrated components [6]: (1) assessment of the availability of opportunities for animals to engage in self-motivated rewarding behaviours, (2) assessment of their actual use of those opportunities and finally, (3) making cautious judgements about the degrees of positive affective engagement the animals may experience, and grading them accordingly.

Examples of grading using these two scales applied to Domain 4 are provided in Section 5.

3.3. The Utility of the 2015 Model for Assessing Animal Welfare

The utility of the Model, summarised here, has been evaluated in detail elsewhere [7,81]. The Model's utility is based on validated scientific foundations of the physical/functional and behavioural indices of negative affects aligned with welfare compromise and positive affects aligned with welfare enhancement. The wide range of affects identified for consideration and the configuration of the domains that was designed specifically to clarify the likely sources of those affects, together enable Model-based welfare assessments to be structured, systematic, comprehensive and coherent. Moreover, seven interacting applications of the Model enable assessors to: (1) specify key general foci for animal welfare management, (2) highlight the foundations of specific welfare management objectives, (3) enable monitoring of responses to specific welfare-focused remedial interventions and/or maintenance activities, (4) identify previously unrecognised features of poor and good welfare, (5) facilitate qualitative grading of specific features of welfare compromise and/or enhancement, (6) enable both prospective and retrospective animal welfare assessments to be conducted and (7) provide adjunct information to support consideration of quality of life (QoL) evaluations in the context of end-of-life decisions. Nevertheless, it is important not to overstate what utilisation of the Model can achieve. Constraints arise through the following factors: (1) different levels of confidence with which particular affects may be inferred to be present in different circumstances, (2) the necessary focus only on the specific affects that can be identified, (3) differing precision with which each affect may be graded and (4) the limits imposed by an inability to determine the relative impacts of different affects when evaluating the notional overall negative–positive affective balance represented by QoL, thereby precluding the possibility of elaborating an all-inclusive QoL metric.

4. The 2020 Five Domains Model: Domains 1, 2 and 3

As outlined above (Section 2.3), Domains 1 to 3 direct attention towards nutritional-, environmental- and health-related survival-critical factors that disrupt or disturb discrete features of the inner stability of the body. Each form of instability has distinctive characteristics that may be identified using measurable physiological, pathophysiological, pathological, clinical and other such indices. Functionally, these indices are detected by specific sensory receptors that send neural impulses to the brain for processing into particular negative affects. Each such negative affect, generated by genetically embedded mechanisms, provides a compelling drive or motivation for the animals to engage in specific behaviours upon which their survival depends (see Section 2.3).

Although the animals would be cognitively aware of each affective experience and the aligned behaviours, they would have little or no ability to stop the behaviours from occurring. For example, the elicitation of behavioural responses to intense breathlessness, pain, nausea and dizziness would likely be almost entirely beyond an animal's control. In contrast, some elements of choice may attend behavioural responses to other affects where the animal needs to identify and/or access locations to undertake a required corrective activity or forms of corrective inactivity. Such corrective activities include seeking water in response to thirst and locating food in response to hunger. On the other hand, corrective inactivity would likely include the seeking of restful isolation in response to debility, weakness and/or sickness. Generally, therefore, agency (i.e., animals' ability to consciously engage in goal-directed behaviours) is not a major part of most behavioural responses to factors noted in Domains 1 to 3. In contrast, agency dominates the behavioural responses considered in Domain 4 (see Section 5).

The brief descriptions of these domains in the 2020 Model provided in Sections 4.1–4.3 include updated or additional examples. Note also the name change of Domain 2 from "Environment" to "Physical Environment". This emphasises that Domain 2 directs attention towards the affective impacts of the largely physical/atmospheric conditions that animals cannot control, and to which they mount or attempt to mount obligatory physiological and pathophysiological responses, often accompanied by supportive behaviours (Section 4.2).

4.1. Domain 1: Nutrition—Imbalances and Opportunities and Their Associated Domain 5 Affects

This domain refers to the water and food available to animals (Figure 2). Intakes may be restricted in animals living outdoors. Examples include the following: when drought depletes natural water sources and limits the available vegetative forage or prey for hunting, when winter temperatures inhibit the growth of vegetation, when deforestation disrupts natural ecosystems, or when uncontrolled reproduction and/or overstocking raise animal numbers above the carrying capacity of rangeland or fenced areas. Poor food quality mainly refers to deficiencies or excesses of trace elements or other essential nutrients and/or inadequate energy and protein contents of plants; for example, those resulting from trace element deficiencies in soils and/or the seasonal growth cycle of grasses, or from the routine feeding of inappropriate diets, such as giving some processed dog food to cats. Low food variety refers to when animals that normally eat varied diets are given the same, albeit nutritious, foods for long periods. Examples include restricting grazing livestock to fenced areas of grass monocultures, long-term feeding of single batches of silage to dairy cows, continuously feeding a dry, nutrient-balanced processed diet to companion dogs, cats or birds and similar continuous feeding of such processed diets to laboratory animals. Negative affects elicited by these inadequacies reflect the nature of the associated welfare compromises (Figure 2).

Such compromises may be avoided or reversed when animals use nutritional opportunities that elicit the positive affects listed in Figure 2. Consideration of such nutritional problems and potential remedial actions are particularly relevant to animals maintained in enclosures that lack the full complement of nutritional conditions for which their species has evolved. This is because practically meeting animals' nutritional requirements in ways that may elicit additional positive affects is the responsibility of the persons charged with their care. In such circumstances, the animals cannot take the required remedial actions themselves.

4.2. Domain 2: Physical Environment—Unavoidable and Enhanced Conditions and Their Associated Domain 5 Affects

This domain focusses attention on the affective impacts of physical and atmospheric conditions to which animals are exposed directly. When the associated affects are negative, the circumstances are categorised as unavoidable physical conditions (Figure 3) because the animals cannot escape from them. For example, in unsuitable indoor housing, these conditions may include space-, floor substrate-, atmospheric-, odorous-, thermal-, noise- and light-related factors, some of which may also lack natural variation. Each such condition is aversive and may elicit identifiable forms of discomfort. Many such conditions may also apply to animals kept outdoors, especially those maintained at high densities or confined in small enclosures; also, those unable to access shelter in cold/wet/windy conditions or shade when hot.

Remedies intended to enhance these ambient conditions can improve the animals' welfare states by enabling them to experience various forms of comfort that may be physical, respiratory, olfactory, thermal, auditory, visual and/or variety-related (Figure 3). Attendant affective experiences may merely be neutral, because specific discomforts are absent. However, this could arguably have permissive effects by minimising unpleasant sensory inputs that would hinder the animals' enjoyment of other experiences (see Reference [7]), for example, noxious odours obscuring attractive smells of food. Attendant affects may also be positive, for example, the pleasurable restfulness of lying on dry, soft, draught-free and hygienic substrates indoors, and the comforting thermal pleasure of basking in the sun.

4.3. Domain 3: Health—Negative and Positive Conditions and Their Associated Domain 5 Affects

This domain focusses attention on the welfare impacts of injury, disease and different levels of physical fitness. Injuries, whether they are acute or chronic, or caused by accidents, invasive husbandry practices, training implements, restrictive devices used to enhance performance, therapeutic surgical procedures, disease-related pathology or poisons, may cause pain that, because of its many different causes, has up to 30 different affective qualities [23]. Acute, chronic or genetic disorders, and persistent functional impairment when spontaneous or assisted recovery is incomplete, may give rise to a range of other negative affects. The character of these affects depends on the organ systems affected and the disease agent, poison and/or pathophysiological processes involved (Figure 4). Extreme overfeeding and underfeeding are included in this domain, and not Domain 1, because the associated pathophysiology may give rise to several of the negative affective experiences noted in Figure 4. Finally, fitness level is included because muscle de-conditioning and bone depletion increase susceptibility to injury and fatigue, the risks of which can be mitigated by levels of exercise that maintain muscle and bone strength (e.g., References [90–92]).

Achieving or maintaining good health and fitness accompanied by a wide range of positive affective experiences (Figure 4) involves using welfare-relevant husbandry practices (Domain 1), facilities design and environmental management (Domain 2) and veterinary attention (Domain 3). It also involves genetic selection for appropriate phenotypes to correct or avoid well-known functional impairments that have dire welfare consequences for production, companion and laboratory animals, and, as recently anticipated, for pest animals [93–96]. These and the previous observations in this section highlight two points: first, that factors included in the first three domains overlap due to the highly integrated functional interactivity within the body operating as an integrated entity (Section 2.1), and second, that these three domains deal mainly with survival-critical conditions and their associated affects (Section 2.3).

5. The 2020 Five Domains Model: Including Human–Animal Interactions in Domain 4

Domain 4, previously named "Behaviour" (Figure 1), has been renamed "Behavioural Interactions" (Figure 5) in order to give greater clarity to its role in the Model. Whereas Domains 1 to 3 mainly focus on animal care-related inputs to welfare, Domain 4 is intended to capture behavioural outputs as indices of animals' perceptions of their external circumstances. More specifically, it highlights the flexible agency-related behaviours animals mount in response to variable, often unpredictable external events and conditions.

Agency is apparent when animals engage in voluntary, self-generated and/or goal-directed behaviours [42,49,53]. More specifically, agency indicates the intrinsic propensity (genetic and/or learned) of an animal to actively engage with its physical, biological and social environment, beyond the degree demanded by its momentary needs, in order to gather knowledge and enhance its skills for future use in responding effectively to varied and novel challenges [42,49,53]. In other words, the exercise of agency involves the cognitive assessment of circumstances in support of animals making mainly conscious choices to behave in particular ways [97,98].

Accordingly, the primary focus of Domain 4 is on behavioural evidence of hindered and/or enhanced expression of agency when animals interact with (1) their environment, (2) other non-human animals and (3) human beings. For these interactions, the aligned affects are largely produced by brain processing of sensory inputs elicited from outside the body. Hence, Domain 4 captures agency-focussed responses to situation-related factors. As noted above (Section 4), this contrasts with the focus of the first three domains on genetically programmed physiological and/or pathophysiological mechanisms inside the body that are specifically directed towards restoring and/or maintaining survival-enhancing internal stability [6,7].

Although the 2015 Model included interactions with the environment and other non-human animals, reference to them was not differentiated structurally. In addition, as mentioned above, human–animal interactions were not included specifically, but were noted as meriting consideration [6]. All three categories are identified explicitly in the 2020 Model.

5.1. Features Common to All Three Categories of Behavioural Interaction

Operationally, the three categories focus on behaviour-based evaluations of affective experiences that animals may have when they direct their attention externally (Figure 5). In terms of impediments, particular negative affects are anticipated when specific agency-related behaviours are absent, or their occurrence is diminished in animals occupying severely restricted, oppressive and/or challenging circumstances, such as those noted in Figure 5. The generation of these affects is considered to result, at least partly, from thwarting of genetically programmed elements of an animal's ethogram, by disabling its engagement in rewarding behaviours and/or by a failure to gain anticipated rewards [44,49,99]. Examples include the following: (1) the daily thwarting of normal long-duration grazing motivation in stabled horses fed with highly concentrated feeds which nevertheless meet their nutritional requirements, (2) the frustrated hunting motivation of canids and felids kept indoors with no suitable substitutes, (3) the frustration of social species such as horses and elephants that are prevented from joining conspecifics engaged in social behaviours, (4) the yearning for company (i.e., loneliness) of isolated individuals of social species kept in separate enclosures and (5) the "separation anxiety" in strongly bonded companion animals due to withdrawal of human company and physical contact.

The opposite of thwarted motivation arises in circumstances that provide opportunities which enhance animals' ability to express agency-related behaviours (Figure 5). Providing such opportunities allows situation-related negative affects to be replaced by positive affects, thereby enabling animals to experience states of "positive affective engagement" [6,7,16,59,60]. Such engagement represents the experience animals may have when they respond to motivations to undertake behaviours that they find rewarding, and it potentially incorporates all of the associated affects that are positive [59,60]. More specifically, enhanced circumstances enable animals to respond to genetically programmed or learned impulses to engage in agency-related behaviours that are linked to affectively positive experiences of anticipation, goal achievement and memory of success [35,37,45,51,53,59,60,63]. Moreover, as the exercise of agency is anticipated to be accompanied by animals having a general sense of being in control of their actions [49,53,63], this would further enhance their feelings of mental security and experiences of positive affective engagement [59,63].

Finally, positive experiences may also arise in ways not directly related to the exercise of agency [6,12]. Examples include the following: (1) herbivores enjoying the pleasant tastes and textures of a variety of feeds delivered to them indoors, just as they would when they self-select and ingest the same feeds while grazing outdoors, (2) the companionable benefits enjoyed by bonded animals yarded together, duplicating those requiring them to actively locate and maintain contact with each other when part of groups on open ranges and (3) humans initiating and maintaining interactive contact with companion animals in the home in ways that provide satisfaction which approximates to that of agency-instigated affiliative interactions between conspecifics in pre-domestication circumstances.

5.2. Animals' Interactions with Humans

The principal focus of human–animal interactions is the impacts of the presence and behaviour of persons as primary causes of animals' behavioural and affective responses. This emphasis includes both animal training and husbandry. In the trained animal, examples are the effects on agency of altered cues and contingencies of learned responses such as ambiguous signals, relentless tactile pressures and altered expectations of reward [100]. Underscoring this, it is well established that: (1) the attitudes, motivation, understanding and skills training of people influence the nature of their behaviour towards animals, (2) it is the impact of their behaviour on the animals that elicits animals' negative and/or positive affective experiences and (3) the nature of the animals' experiences may be inferred from their behavioural and physiological responses (e.g., References [12,29,55,60,87,101–115]).

Figure 5 provides examples of salient human characteristics, subdivided according to attitude, voice, aptitude and handling/control, as well as examples of animals' impeded or enhanced agency-related behaviours and their aligned affective experiences. The examples are just that. They are neither definitive nor exhaustive, nor should they be generalised to all animals. Also, the listed negative and positive human attributes and animals' affective experiences are intended to indicate possible negative-to-positive ranges, thereby facilitating consideration of these factors at and between these extremes. As with all other animal-based examples provided in Figure 5 (also in Figures 2–4), users of the Model should evaluate them with regard to any unique behavioural, biological and ecological features of the species in question, together with the precise circumstances of the animals being considered. The purpose here is for Model users to decide whether any named human behaviours and/or induced animal behaviours or affects should be deleted, retained or modified, or whether others should be added.

As an adjunct to Figure 5, Figure 6 lists some of the general circumstances in which Model use could include assessments of the impacts on animals of specific negative and/or positive features of human proximity and/or behaviour. It also provides examples of activities or occupations where those circumstances may apply.

More than one of these general circumstances may be applicable to particular examples of human–animal interactions if they develop over time or when different interactions occur in sequence. Also, the examples in Figure 6 are not exhaustive. Rather, their purpose, as with the examples in Figures 2–5, is to highlight a range of factors that the Model may be used to evaluate. We encourage the introduction of other examples that may be more applicable to the circumstances and the species-specific attributes of the animals being considered.

5.3. Grading the Negative and Positive Impacts of Humans in Their Interactions with Animals

The grading of welfare impacts in Domain 4 is focussed on the observable behaviour of animals during and following their interactive engagement with (1) different features of their environment, (2) other non-human animals and (3) humans in their vicinity (Section 5.2). Of course, such impact grading must also include any germane elements of the wider circumstances of the animals as revealed by all other aspects of the Model assessment, captured via Domains 1 to 3 (Sections 4.1–4.3).

Human-animal interactions likely to generate negative affects [examples]
Persons near animals that have had little or no prior human contact [*Animals:* rangeland, free-roaming and feral animals; wild-caught fish for display and other wildlife caught for use as pets.]
Persons whose presence adds to already threatening circumstances [*Wildlife managers:* trap-kill, trap-mark-release, capture-relocate; closely confined with no refuge; hands-on zoo visitor or tourist events. *Livestock handlers:* farmhands, transport drivers, sale yard staff, slaughterhouse workers. *Veterinary care teams:* veterinarians, veterinary nurses, animal attendants and owners.]
Persons whose current actions are directly unpleasant, threatening and/or noxious [*Actions:* psychological/physical abuse; serious mistreatment or neglect; physical restraint for aversive management or therapeutic procedures; aversive training methods; separation from dependently bonded companion animals; some veterinarians; riders whipping tired horses in sport.]
Persons whose prior actions are remembered as being aversive or noxious [*Persons:* intentionally cruel persons, unskilled trainers, unskilled animal handlers, stockpersons who apply routine noxious procedures, some researchers, some veterinarians, some farriers.]
Bonded humans whose actions cause unintended harm [*Actions:* affectionate displays seen as threatening by the animal; owners absent from bonded pets for long periods; owners delaying efficacious therapies; delayed end-of-life decisions for animals with compromised welfare.]
Human-animal interactions likely to generate positive affects [examples]
The companionable presence of persons who provide company and feelings of safety [*Persons:* Owners and/or caregivers whose animals are closely bonded to them, including companion, recreational, hobby farm, service, disability, breeder and other animals.]
Persons who provide preferred foods, tactile contacts and/or training reinforcements [*Persons:* companion animal owners, animal care staff; trainers using positive reinforcements; zoo staff using food enrichments.]
Persons participating in enjoyable routine activities [*Activities:* games, daily exercise, regular training.]
Persons participating in engagingly variable activities [*Activities:* diverse daily service functions, training schedules and/or opportunities for new experiences.]
The calming presence of familiar persons in threatening circumstances [*Actions:* hands-on gentling by persons strongly bonded to the animals.]
Persons acting to end periods of deprivation, inhibition or harm [*Activities:* delivering water, food, company and liberty from confinement.]

Figure 6. Some general circumstances in which the presence of humans at a distance, close to or in direct contact with animals may lead the animals to have negative or positive affective experiences, and some specific examples of those circumstances. The examples provide an indication of the negative-to-positive range of human–animal interactions. These, when considered together with the negative-to-positive range of influential human attributes illustrated in Figure 5, are provided to help Model users to evaluate in more detail the impacts of interactions at and between these extremes.

More specifically, the grading of the impacts of the proximity and animal-centred behaviour of humans employs the same two scales as for all other features of Domain 4 (Figure 5), as also of Domains 1 to 3 (Figures 2–4). Hence, it uses the five-tier scale (A to E) for negative impacts and the four-tier scale (0, +, ++, +++) for positive impacts (Section 3.2). However, the examples of graded negative impacts in Figure 7 and graded positive impacts in Figure 8 highlight an important difference.

Nature of Human-Animal Interaction	A: None	B: Low	C: Mild to Moderate	D: Marked to Severe	E: Very Severe
	Lower ← Negative Affective Impact → Higher				
Wildlife handling/control: • Prior human contact • Fight, flight, freeze, appeasement behaviours • Inferred fear	Calm hand-rearing from birth None None	Regular non-aversive handling Very low to low Very low to low	Regular visual contact, no handling Moderate to marked Moderate to marked		No visual or physical contact Extreme Extreme
Livestock handling/control: • Prior human contact through gradual habituation or during sensitive socialisation periods • Restraint level required • Cortisol + behavioural responses and inferred fear	Calm taming and training; fully compliant animals Gentle handling No responses or fear	Feedlot animals with regular contact Light restraint Low responses and fear	Paddock animals with some contact Firm restraint Moderate responses and fear	Rangeland animals with some contact Strong restraint Marked responses and fear	Feral/wild animals with no contact Very strong restraint Extreme responses and fear
Aversive training of companion animals: • General features of training • Fight, flight, freeze, appeasement behaviours • Fear of trainer and others • Response to trainer	Gentle, calm methods No such responses No fear evident Engaged, compliant, at ease	Punishment sometimes used Mild such responses Mild fear sometimes evident Sometimes non-compliant, mostly at ease and compliant	Regular hitting and shouting Responses apparent when trainer is present Moderate fear overall; marked fear when trainer is present Distracted, nervous, anxious, often non-compliant, sometimes aggressive		Brutal methods Extreme responses, often withdrawn Extreme fear of trainer & others Terrified, panicked; aggressive, shutdown, non-responsive
Pushing performance animals to their physiological/physical limits: e.g., using persistent unassisted urging, whips, spurs and/or drugs	Animals compliant; briefly exercised below maximum levels using unassisted urging; little if any fatigue; rapid return to resting state	Animals compliant; briefly exercised at maximum using persistent unassisted urging; some fatigue; return to resting state somewhat delayed	Animals non-compliant due to persistent use of contradictory aversive stimuli; escalation of injurious force; slow decline of pain and fear to resting levels; markedly fatigued; delayed recovery to resting state		Brutal methods lead to extreme withdrawal; panic, terror; pain from injury; bone fractures when fatigued animals misstep; very slow recovery or euthanasia
Denying animals the expression of natural behaviours by using: • *Horses:* bits, tongue-ties, tight nosebands • *Dogs:* anti-barking muzzles	*Horses and dogs:* No restrictive devices used; animals freely express natural behaviours No negative affects			*Horses:* Two devices used with marked pressure, or one device used with severe pressure Severe pain experienced	*Horses and dogs:* All three devices used with marked to severe pressure; or one or two devices used with extreme pressure Very severe pain experienced
Strong human-animal bond: • Animals reliant on humans as their sole companions • Negative affects experienced	Consistent close proximity and affiliative actions maintain strong mutual bonds No negative affects	Mutual bond strong, but animal left alone due to part-time work away from home Transient loneliness and boredom	Strongly bonded animal often left alone due to full-time work away from home Repeated more sustained loneliness and anxiety	Strongly bonded animals left for several months in small kennels where indifferent staff routinely maintain hygiene, food and water and do not provide comforting physical contact Severe anxiety, fear, loneliness and depression; inappetence-induced hunger and physical weakness	

Figure 7. Examples of graded negative affective impacts due to different human interactions with animals of different types and in different situations. For each type of interaction, grades indicated in each row relate to variations in relevant factors of the interaction, such as the animal's prior contact with humans or the training regime. Also noted for each sub-scenario is the degree to which behavioural and/or physiological indicators of the affective experience are expressed by the animal, as well as the intensity of specific inferred negative affects, e.g., fear. The approach here is therefore similar to the grading of other negative impacts.

Nature of Human-Animal Interaction	Lower ⇐ Positive Affective Impact ⇒ Higher			
	None (0)	Low-level (+)	Mid-level (++)	High-level (+++)
Frequency of affiliative contacts	All contact neutral and brief, but not threatening or aversive	Irregular or brief daily interactively engaged contact	Regular daily interactively engaged contact for moderate periods	Companionable interactions for most of each day
Variety of affiliative contacts	Routine daily maintenance activities executed without engagement	Limited variety of shared engaging activities	Moderate variety of shared engaging activities	Wide variety of shared engaging activities
Duration of affiliative contacts	Too brief to approximate to shared activities with members of the conspecific social group	Limited duration of shared engaging activities	Moderate duration of shared engaging activities	Optimal duration that approximates to shared activities with the conspecific social group
Form of affiliative contacts	Too different to approximate to shared activities with members of the conspecific social group	Some similarity with shared congenial activities of the conspecific social group	Moderate similarity with shared congenial activities of the conspecific social group	Optimal form that approximates to shared congenial activities with the conspecific social group. e.g., optimal reciprocal, i.e., not one-sided, bonded congenial contact
Maximum numerical score	0	4	8	12

Figure 8. Examples of relative positive affective impacts on animals due to human interactions graded separately according to the frequency, variety, duration and form of congenial contacts. As these four features interact, they all need to be graded for each situation, and their grades amalgamated into an overall grade. For this purpose, a numerical score is applied to each feature in each column. Note that this is a numerical aid to the qualitative assessment of positive impacts. If any feature receives a zero score, none of the other levels apply and the overall score is zero. The minimum overall score above zero is 4 (1 for each feature), an intermediate score is 8 (2 for each feature) and the maximum is 12 (3 for each feature). The range of possible overall scores above zero in each situation would therefore be 4 to 12. As each feature is graded qualitatively before amalgamation, each overall numerical score is merely a guide for prospectively or retrospectively comparing outcomes of proposed or completed changes by undertaking a succession of such assessments. Note that such comparisons within specific situations is qualitatively meaningful, whereas such comparisons between different situations is not.

The grading of negative impacts is based on assessments of the physiological, behavioural and clinical impacts of human proximity and behaviour on the animals. Thus, the grades in each row of Figure 7 represent a separate relative assessment of variants of particular situations, for example, behavioural responses of wildlife that have had different levels of prior exposure to humans.

In contrast, illustrated in Figure 8 is the grading of the positive impacts of affiliative human–animal interactions, where four key features of human-initiated interactions are emphasised; namely, frequency, variety, duration and form. As these four features also interact, they all need to be graded for each situation, and their grades amalgamated into an overall grade (see Figure 8 for more details). What is presented, therefore, is a means of more thoroughly assessing the human contributions to positive impacts in a wide range of situations. As these assessments are qualitative, the overall grades for different situations cannot be compared meaningfully, but repeated assessments of the same system to detect negative or positive changes would be meaningful (Figure 8).

Although devised here for the assessment of positive human-animal interactions, reference to the four key interactive features of frequency, variety, duration and form of interaction noted above also has application to assessments of positive animal-to-environment and animal-to-animal interactions. We anticipate the need to explore the influences, relevance and consequences of various behavioural conditioning techniques (i.e., training) on the welfare of animals, as viewed through the Five Domains lens. In particular, there will be value in assessing interactions between the outcomes of different modes of learning (associative and non-associative), considered against the backdrop of the animals' evolved capacities to function and behave in species-specific ways, i.e., in relation to their telos (see References [116,117]).

6. Conclusions

Renaming Domain 4 "Behavioural Interactions" (previously "Behaviour"), highlights the inherent capability of sentient animals to consciously self-select goal-directed behaviours when interacting with key features of their environment, with other non-human animals and with humans. When they achieve their selected goals, they may experience one or more of a wide range of welfare-enhancing positive affects (Figure 5). These are rewarding and provide motivation to again engage in the selected behaviours, subjectively experienced as different forms of 'positive affective engagement' [59]. In contrast, if the external circumstances hinder animals from engaging in behaviours that they would find rewarding, they may experience one or more of a range of unpleasant and demotivating negative affects (Figure 5) [6,7]. Animals' agency-related interactions with their environment and with other non-human animals in their environment have been described in detail previously (for References see [6,7,12,16,21,51,59,60]). However, humans also feature as influential in animals' external circumstances, and their interactive behaviour towards animals has the potential to elicit welfare-enhancing positive affects or welfare-compromising negative affects. The Five Domains Model as reconfigured here now provides an explicit means to effectively and systematically evaluate the animal welfare implications of a wide range of human–animal interactions. This extension of the Model is therefore recommended to readers for their consideration and use.

To assist users of the 2020 Model, we have prepared a freely available online summary poster, which combines Figures 2–5 [118].

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Seasonal Variation in Habitat Selection by Free-Ranging Feral Horses Within Alberta's Forest Reserve

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Abstract

Little is known about habitat selection by free-ranging feral horses in Montane environments, including how horse use may vary seasonally throughout the year. We tracked four global positioning system collared horses in four separate harems between November 2008 and October 2010 for a portion of the Rocky Mountain Forest Reserve in southwest Alberta, Canada. We assessed seasonal habitat selection for the study period by combining locational data with landscape data (including vegetation types) in an information theoretic framework. Home ranges for horses varied from 12.4 to 90 km^2 and were confined to local watersheds. Horses selected most for lowland grasslands across all seasons, with shrublands increasingly selected in spring and summer. Harvested conifer forests were only selected by horses during winter. Resource selection functions indicated that in addition to vegetation type, horses were selecting for a variety of habitat characteristics (i.e., distance to forest and solar radiation), while water availability, topographic accessibility, and disturbance features (e.g., distance to roads, recreational trails, and seismic lines associated with energy exploration) had little or no influence on horse selection. Overall, horses demonstrated selection for habitats covering 14% of the study area while avoiding 42% of habitats: remaining areas were used in proportion to their availability. Concentration of horse use within sparse vegetation types (grassland and shrubland), particularly during one or more times of the year, help identify critical horse habitat including areas where multiple, overlapping land uses interact on public land.

Key Words: disturbance corridors, geographic information system, landscape features, thermal protection, vegetation type, water availability

INTRODUCTION

Following the Pleistocene mega-faunal extinction, domestic horses (*Equus ferus callabus*) were introduced to North America in the 1500s (Lever 1985; Singer 2005). Feral herds of free-ranging horses now occupy large parts of the western United States and portions of Canada. Horse management has received significant attention and led to the implementation of protective legislation in both the United States (Bureau of Land Management 2011) and recently, select regions of Canada (Government of Saskatchewan 2009). This political intervention has led to prominent increases in horse populations and in some instances to declines in herd health and range condition (Humane Society of the United States 2005).

In the foothills of southwest Alberta, free-ranging feral horses have been present since the early 1900s (Government of Alberta 2011). While many of these horse populations originated from unwanted and released draught animals that evaded capture attempts in the 1920s, they have been supplemented by released or escaped individuals, as evidenced by the presence of horses with brands. Feral horse populations in this region have increased from approximately 700 head in 2009 to over 1 000 head in 2011 (Alberta Sustainable Resource Development [ASRD], unpublished data). These numbers represent numerous harems containing one stallion with multiple mares and foals (McCort 1984; Linklater et al. 1999). In Alberta, harems typically consist of 3 to 17 animals (Salter and Hudson 1982). Increases in the horse population, coupled with declines in the availability of grassland habitat, have raised concerns over the long-term conservation of horses and their primary habitats.

Habitat selection and use by herbivores is influenced by many factors (Anderson 2010). In addition to population size, the spatial and temporal patterns of habitat use by herbivores are important considerations (Senft et al. 1987), particularly in environments with strong seasonal variability such as northern temperate forests. In the predominantly forested foothills of Alberta, horses prefer open grasslands and shrublands during summer (Girard et al. 2013), which may reflect their preference for herbaceous vegetation over browse (Salter and Hudson 1979). Horses are also known to select areas with greater biomass to enhance foraging efficiency (Fleurance et al. 2009). In heavily forested environments that lack grasslands, horses select disturbed areas such as road side edges and seismic lines (i.e., linear clearings used for energy exploration) where grass production is high (Irving 2001). Although low water

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availability will reduce habitat selection by horses (Stevens 1988), the effects of water availability on horse use in Alberta appear to be minimal. Responses vary from positive associations in summer (Girard et al. 2013) to no particular influence throughout the year (Salter and Hudson 1979).

Terrain is also known to influence habitat selection, with horses more likely to occupy flat pastures or gently sloping ridgetops (Ganskopp and Vavra 1987). Accessibility plays a key role in regulating animal movement due to associated effects on energy expenditure (Senft et al. 1987). In rugged topography or dense vegetation, the presence of roads and trails can increase accessibility, thereby increasing use of habitats in close proximity to these corridors. However, increased human presence along roads and trails (i.e., motor vehicles, recreational vehicles, hikers, etc.) may decrease use of these habitats, as it has for wildlife (Laliberte and Ripple 2004). The net impact of corridors on feral horses in forested rangelands remains unknown. Finally, animal exposure to habitats at a young age can positively influence future habitat use (Bailey et al. 1996; Launchbaugh and Howery 2005). Harems remain loyal to a home range once established and frequently follow examples set by older animals (McCort 1984).

The eastern slopes of the Rocky Mountains contain high spatial and temporal variability in habitat availability for free-ranging herbivores (Hebblewhite 2005). Given the importance of grasslands to biodiversity in the region, a greater understanding is needed of habitat selection and use by free-ranging feral horses. The objective of this study was to use global positioning system (GPS) technology to 1) quantify habitat selection by feral horses within a foothill landscape of southwest Alberta, including seasonal changes in selection, and 2) evaluate potential mechanisms influencing spatio-temporal variation in habitat selection by feral horses, including the role of vegetation type, topography, water availability, travel corridors, distance to forest, and thermal characteristics.

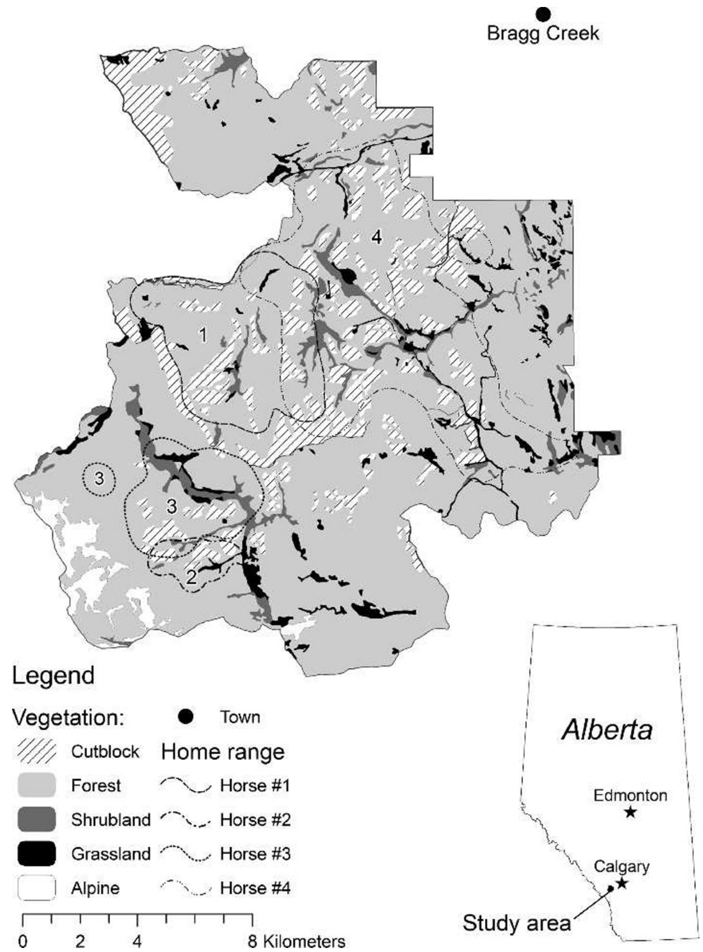

Figure 1. Distribution of vegetation types within the study area and associated home ranges of the four horses studied, within the Rocky Mountain Natural Region of southwest Alberta, Canada.

METHODS

Study Area

Feral horses were studied in a 202-km^2 area west of Bragg Creek, Alberta, in and around the McLean Creek Forest Land Use Zone of the Rocky Mountain Forest Reserve (RMFR; Fig. 1). The RMFR is an area of public land managed for multiple uses, including wildlife management, forest harvest, cattle grazing, and recreation and watershed protection, among others. Landscapes in the area fall within the Montane and Subalpine Natural Subregions, with elevations ranging from 1 341 to 2 331 m (Natural Regions Committee 2006). Vegetation consists of a mosaic of sparse grasslands and riparian shrublands along valley bottoms, with uplands comprising occasional deciduous or mixedwood forests, widespread conifer forests, and numerous harvested conifer forests, or cutblocks (ASRD 2005). The area comprises 69% conifer forest, 13% conifer cutblocks, 4% mixedwood forest, 4% shrubland, and 4% grassland, with the remainder made up of water, rock (including alpine), or heavily disturbed areas. Plant communities vary widely in herbage production, but generally follow the ranking of: grasslands > shrublands > conifer cutblocks > mixedwood forests > conifer forests (ASRD 2005).

Weather varies greatly among seasons, with daily mean temperatures at the nearby Elbow Ranger Station ranging from –9°C in January to 12°C in July and August (Environment Canada 2010). Mean annual precipitation for the area is 644 mm, with most falling as rain between 1 May and 31 August. Annual precipitation for both years of the study remained near normal, although seasonal patterns of precipitation differed between years (Fig. 2).

Habitat Use by Horses

Approximately 131 feral horses, distributed among 11 harems, are found in the study area. Four randomly selected mares from four different harems were tranquilized from a helicopter and fitted with GPS collars by Alberta Environment and Sustainable Resource Development (AESRD) staff under supervision of a practicing veterinarian in October of 2008. Horses ranged from 3 to 7 yr in age, were of a medium frame size (approximately 12.2 to 14.2 hands high), and were healthy and representative of the majority of mares in the herd. Three of the four mares were pregnant at the time of collaring. Harem sizes ranged from 9 to 27 in size at the time of collaring. Only mares were collared as they are less likely to be involved in dominance

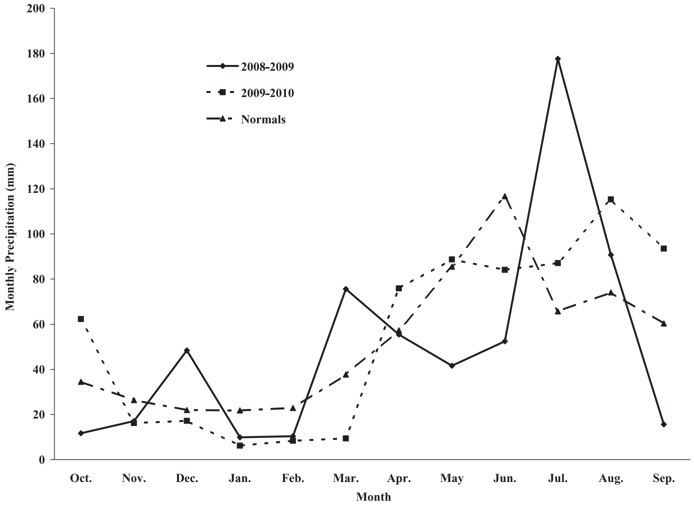

Figure 2. Actual (2009 and 2010) and long-term (30-yr) mean monthly precipitation for the study area according to the Elbow River Ranger Weather Station.

fighting and are more likely than stallions to remain within the same harem.

Lotek 7000 series GPS collars were programmed to record GPS locations once every hour for a 2-yr period between 28 October 2008 and 8 October 2010 for one mare, and until 25 October 2010 for the remaining three mares. Collars recorded the date and time, location (elevation, latitude, and longitude), dilution of precision (DOP), ambient temperature, number of satellites used to obtain a fix, viability of the fix, and the type of fix (2D or 3D; Lotek Wireless Inc. 2011). No problems were encountered with the collars, and data were remotely downloaded every 6 mo. Collars weighed approximately 1.25 kg and did not appear to interfere with routine horse behavior.

Data on feral horse locations were entered into a geographic information system (GIS) using ArcGIS 9.3 (ESRI 2009) and converted to Universal Transverse Mercator format. Datasets were initially screened for errors caused by obvious incorrect fixes (i.e., points outside the study area) or high DOP (>6), leading to the removal of 9.6% of observations because they were considered inaccurate (D'Eon et al. 2002). Conifer forests were most likely to experience poor satellite reception due to interference with tall trees (Rempel and Rodgers 1997; Dussault et al. 1999).

Landscape Factors of Habitat Selection

Spatial databases describing different landscape features for the study area (Table 1) were obtained from provincial digital data archives (AESRD). Vegetation types were defined by the Alberta Vegetation Inventory (AVI), which is based on photo-interpreted vector polygons of uniform vegetation age, structure, and composition. AVI maps were grouped into the following five broad categories for electivity analyses of vegetation selection: conifer forest, conifer cutblocks, mixedwood forests, grasslands, and riparian shrublands.

For the subsequent analysis investigating both categorical and continuous factors influencing habitat selection, conifer and mixedwood forest were assessed both individually and together in a single combined "forest" variable. In addition, a topographic ruggedness index (TRI) and an index of solar radiation exposure were estimated for the study area using a 25-m digital elevation model. TRI was estimated using an ArcScript by Riley et al. (1999) that assesses changes in elevation between adjacent grid (25 m) cells. Solar radiation exposure was calculated for both diffuse and global solar radiation for the first official day of spring (21 March) using an ArcScript based on the equations from Kumar et al. (1997). Finally, the "near" function in ArcMap 9.3 was used to generate distances between horse locations and the various landscape features, including forest cover, water, and roads or trails.

Home Ranges and Sampling of Habitat Availability

Home ranges are areas where animals perform normal activities and spend the majority of their time (Burt 1943). Home ranges were created for each collared horse to determine the availability of habitat and landscape features. As home ranges showed little interannual variation, a single home range was developed for each animal for the entire study period (Fig. 1). Home ranges from different animals were independent based on visual assessment, as home ranges typically followed watershed boundaries. Moreover, collared horses were never found together during the study, with limited overlap in home ranges (Fig. 1), suggesting collared horses and their associated harems remained independent.

To define home ranges, we used kernel density methods, a nonparametric statistical approach for estimating probability densities from a set of locational points (Rodgers and Kie 2010). Kernel home ranges were created using the Home Range Tools developed by Rodgers et al. (2007) in ArcMap 9.3.1 (ESRI 2009). As recommended by Blundell et al. (2001), fixed kernel distributions with the reference bandwidth were used to develop home ranges with 95% use polygons. A 95% kernel home range was used for all analyses to account for the majority of horse activities. Since there was a 1-hr time lag between successive GPS location points, spatial autocorrelation was likely present in the data. However, work done by de Solla et al. (1999) found that an increased number of data points improved spatial accuracy and precision; therefore, the entire corrected data set was used to ensure robust home range development.

Random points were generated at a density of one location per hectare to estimate the availability of habitats for each horse. Random locations were assessed for the same landscape features as horse locations, thereby allowing comparison of used and available spatial data for each horse.

Habitat Electivity and Resource Selection Function Analysis

Resource selection functions (RSFs) quantify how animals select areas of the landscape (Manly et al. 2002). We used a type III study design (Manly et al. 2002) where selection of used vs. available resources was assessed specific to each horse. Used resources were defined seasonally from horse location information (i.e., the proportion of total observations within each habitat), while available resources were generated for each horse within individual home ranges.

Vegetation use data (i.e., horse point locations) were compared with vegetation type availability (i.e., random

Table 1. Description of habitat themes and associated variables developed in ArcGIS 9.3 for use in the assessment of feral horse resource selection.

Theme[1]	Variable	Description
Vegetation	Avoided	Habitat polygon with electivity < 0. 1 = use, 0 = nonuse
Type	Neutral	Habitat polygon with electivity not different from 0. 1 = use, 0 = nonuse
	Selected	Habitat polygon with electivity > 0. 1 = use, 0 = nonuse
Water and	D. water	Distance from horse/random points to nearest source of water (100 m)
Topography	Elevation[2]	Elevation above sea level ranging from 1 341 to 2 330 m
Access and	D. roads/trails	Distance from horse/random points to nearest road or trail (100 m)
Disturbance	D. seismic lines	Distance from horse/random points to the nearest cutline (100 m)
Thermal	D. mixedwood	Distance from horse/random points to the nearest mixedwood forest (100 m)
	D. conifer	Distance from horses/random data points to the nearest conifer forest (100 m)
	D. any forest	Distance from horses/random data points to nearest forest (100 m)
	TRI[2,3]	Terrain ruggedness index, increasing values indicate increasing roughness
	DSR[2]	Diffuse solar radiation. Measure of scattered wavelengths on March 21
	GSR[2]	Global solar radiation. Measure of shortwave + diffuse radiation
Interactions	D. water × TRI	Combination of distance to water and ruggedness
	D. water × elevation	Combination of distance to water and elevation
	TRI × elevation	Combination of elevation and ruggedness

[1]See text for a detailed explanation of themes and associated variables.
[2]Indicates raster data.
[3]TRI indicates topographic ruggedness index; DSR, diffuse solar radiation; GSR, global solar radiation.

points) using Ivlev's Electivity Index (EI; Ivlev 1961; see Equation 1) to determine horse selection for each vegetation type.

$$EI_{\text{veg type ``x''}} = (\%\text{horse use in ``x''} - \%\text{ of ``x'' available})/(\%\text{horse use in ``x''} + \%\text{ of ``x'' available}) \quad [1]$$

Electivity data indicated those vegetation types that were selected ($EI > 0$), avoided ($EI < 0$), or neutral (i.e., habitat was occupied in the same proportion as available on the landscape; $EI = 0$). Electivities were calculated separately for each horse, and examined for year, season, and time of day effects. Differences in electivity among vegetation types were then tested in SAS 9.2 with the residual maximum likelihood method, incorporating individual horse as a random effect (Gillies et al. 2006).

Following the electivity analyses of vegetation types, RSF analyses (i.e., information theoretics) were used to evaluate the relative influence of both categorical and continuous landscape features on horse habitat selection. Separate RSFs were developed for winter, spring, summer, and fall seasons over both years, as preliminary analysis of the habitat data revealed marked differences in electivity between seasons, but not years. Cut-off dates between seasons were established from combinations of expected changes in plant growth and associated forage availability based on known changes in plant phenology, snow cover, etc. Using these criteria, the winter season was defined as 1 November to 31 March and coincided with the period of snow cover. Spring was from 1 April to 15 May, representing the short transition from vegetation dormancy through initial green-up. Summer was defined as 16 May to 15 September, and included the growing season and period of greatest herbage production and forage availability. Finally, fall was defined as 16 September to 31 October, coincident with rapid plant senescence before snow fall reduces forage accessibility. Analysis comparing horse distributions between day and night revealed no clear diurnal patterns; thus, no further division based on time of day was considered.

In preparation for the RSF analysis, used and available spatial locations, along with all vegetation type and other habitat (i.e., landscape) variables, were combined to create a dataset for each horse. Used data points were set to "1," while those available were set to "0." Variables used for resource selection (see Table 1) were initially examined for redundancy using Pearson's correlations with Proc CORR in SAS 9.2. Variables correlated at $r > 0.7$ across all horses were considered redundant and removed, leaving one variable per group. However, variables were retained when at least one animal did not exhibit correlation prior to data combination. The diffuse solar radiation and elevation by ruggedness interaction were both correlated with ruggedness. Ruggedness was retained because it was considered representative of many environmental variables. Similarly, the distance to water by elevation interaction was correlated with distance to water, with the latter retained because of its ease of measurement and interpretation.

As a first step in the RSF, variables were divided into themes representing different a priori hypothesized factors influencing use patterns by feral horses (Table 1), which reduced the number of variables for final comparison in a hierarchical manner. To determine the most representative variables from each theme the –2 log likelihood (–2LL) was obtained using Laplace Approximation with horse as a random effect in Proc GLIMMIX in SAS 9.2 (Gillies et al. 2006). The –2LL was used to generate a pseudo R^2 (goodness-of-fit) for each model to compare the percentage of deviance explained by all models in comparison to the null (Windmeijer 1995; Cameron and Windmeijer 1997; see Equation [2]).

$$\text{McFadden's pseudo } R^2 = 1 - (\text{log likelihood candidate model}/\text{log likelihood null model}) \quad [2]$$

Within each theme, the model that best explained deviance in horse use was selected. Usually this was the model with the

Table 2. Mean electivity for various vegetation types by feral horses in the Rocky Mountain Forest Reserve of Alberta from October 2008 through October 2010. Electivities with a * indicate those that horses either preferred (> 0) or avoided (< 0), at $P < 0.05$. Pooled standard error ± 0.15 across all treatments.

Vegetation type	Winter (1 November–March 31)	Spring (1 April–May 15)	Summer (16 May–September 15)	Fall (16 September–31 October)
Conifer	−0.444* AB[1] c[2]	−0.618* B c	−0.300* A c	−0.19* A b
Cutblock	0.328* A a	−0.102 B b	0.046 AB b	0.073 AB ab
Grassland	0.506* A a	0.718* A a	0.602* A a	0.226* B a
Mixedwood	−0.053 A b	0.190* A b	0.013 A bc	−0.046 A ab
Shrubland	−0.005 A b	0.195* A b	0.192* A b	−0.096 A b

[1]Seasonal means within a row with different uppercase letters differ, $P < 0.05$.
[2]Vegetation type means within a column with different lowercase letters differ, $P < 0.05$.

greatest percent deviance explained, with the condition that increasing the number of variables required an increase of at least 1% deviance per variable. Where no model had an explanatory power greater than 1%, the best model was chosen to move forward to represent that theme in the final model testing. Model selection was completed separately for each season. Finally, additional models were created treating avoided, neutral, or selected (ANS) vegetation types (i.e., core selection) as a null model following the same process outlined above.

Once the best model from each theme was identified, these models were combined in an additive fashion and run through Proc GLIMMIX to determine the final model that best accounted for overall patterns of horse selection. The first model used the theme with the greatest explanatory value from the previous stage. Themes were added and tested in descending fashion, and carried forward to the next test provided they yielded a 1% increase in pseudo R^2. This was done for each season to generate the final models and variables for inclusion in the RSFs. Final RSFs (Manly et al. 2002) were developed to describe the relationships between horses and various significant landscape characteristics (see Equation [3]).

$$RSF = \exp(\beta_1 x_1 + \ldots + \beta_p x_p) \quad [3]$$

Finally, beta (β) coefficients were obtained from the Proc GLIMMIX (SAS Institute 2007) output used to produce the –2LL, and the RSFs used to predict habitat selection across the study area representing the likelihood of horse presence for each season. This was done using the Map Algebra function in ArcMap 9.3 where the betas from the logistic model were used to predict local habitat selection based on landscape values for each 25-m pixel.

RESULTS

Home Ranges and Vegetation Electivity

Kernel home range analysis indicated that horses occupied different areas of the study area and had varying home range size. The 95% kernel home ranges of the four horses ranged from 12.4 to 90.0 km^2 (mean=48.4 km^2). Home range sizes closely followed individual watershed boundaries, rarely extending into adjacent watersheds, but were also highly correlated with the initial size of harems for each collared horse (r=0.97).

Horse electivity for different vegetation types varied within individual seasons (Table 2). In winter, horses selected conifer cutblocks and grasslands ($P < 0.05$). In spring, lowland grasslands, mixedwood forests, and riparian shrublands were selected, a pattern that continued through summer for the two nonforested habitats. During fall, horses selected lowland grasslands and exhibited similar electivity for cutblocks.

Selection of individual vegetation types by horses also varied seasonally (Table 2). For example, grasslands were strongly selected in every season but remained lower during fall than at other times of the year ($P < 0.05$). Riparian shrublands were selected in spring and summer. Although conifer forests were avoided in all seasons, this habitat was avoided most during spring. Selection for cutblocks occurred only in winter, with this vegetation type being neutral in all other seasons. A similar pattern was evident for mixedwood forests with selection only in spring.

Resource Selection Functions

Comparison of the *a priori* models within individual themes indicated that the same variables or variable combinations explained the majority of deviance in horse presence across the study area during winter, spring, and summer (Table 3). The core ANS model, representing the vegetation theme, was carried forward to all RSF models as it represented our null model of general selection for vegetation types. For the water and topography theme, ruggedness was selected as the most important factor. Within the disturbance theme, distance to roads and trails was selected as the most important factor although it explained little variation (< 1%) in habitat selection. The model that explained the most deviance in the thermal theme was distance to both forest types (mixedwood and conifer), in combination with solar radiation. This model also explained more deviation in horse selection than all other themes (Table 3). Comparative models between seasons were generally consistent in variable selection within themes, with one notable exception: within the disturbance theme, the fall model with roads and trails in combination with seismic lines explained more deviance than roads and trails alone (Table 3). When ranking individual themes (hypothesized factors) their order of importance was: thermal > habitat > water and topography > disturbance. This ranking was consistent across all seasons.

In the final analysis (i.e., model combination across themes) of winter horse data, the model that explained the most deviance was the "thermal+habitat" model at 21.3% (Table 4).

Table 3. Summary results depicting comparative model strength linking feral horse observations from global positioning system telemetry data collected during winter (1 November–31 March), spring (1 April–15 May), summer (16 May–15 September), and fall (16 September–31 October) 2009 and 2010, and various landscape attributes. Bolded and italicized components indicate the leading model in a theme, and which were carried forward into the final assessment.

Theme	Component	K[1]	R^2 Winter	Spring	Summer	Fall
Null		1	0.00[2]	0.00	0.00	0.00
Vegetation type	ANS[3]	3	***11.19***	***17.18***	***7.92***	***3.19***
Water and topography	D. water	2	0.01	0.69	0.00	0.03
	TRI	2	***3.55***	***1.35***	***3.45***	***1.77***
	D. water × TRI	2	0.68	1.22	0.89	0.18
	D. water + TRI	3	3.57	2.06	3.45	1.79
	D. water + TRI + D. water × TRI	4	3.92	2.15	3.69	2.35
Disturbance	D. roads/trails	2	***0.25***	***1.12***	***0.91***	0.42
	D. seismic lines	2	0.01	0.33	0.08	0.88
	D. roads/trails + D. seismic line	3	0.29	1.27	0.94	***1.18***
Thermal	D. any forest	2	5.90	6.31	3.84	1.67
	D. conifer	2	3.12	3.52	2.10	1.16
	D. mixedwood	2	7.01	7.25	7.83	6.41
	GSR	2	3.31	6.87	2.05	1.36
	D. conifer + D. mixedwood	3	11.77	12.92	11.94	8.94
	D. conifer + GSR	3	6.30	10.70	4.03	2.49
	D. mixedwood + GSR	3	9.89	13.24	9.67	7.53
	D. any forest + GSR	3	9.04	13.49	5.79	2.99
	D. mixedwood + D. conifer + GSR	4	***14.44***	***19.30***	***13.62***	***10.03***

[1]Indicates the number of parameters used.
[2]McFadden's pseudo R^2 goodness-of-fit measure.
[3]ANS indicates avoided, neutral, or selected; TRI, topographic ruggedness index; GSR, global solar radiation.

Variables included in the final winter model were distance to conifer and distance to mixedwood forests (i.e., uncombined), solar radiation, and vegetation type (ANS) selection. In the final spring analysis, the leading model was "thermal+habitat+disturbance," explaining 31.5% of deviation in horse distribution (Table 4). Variables included in the spring model were the same as winter, with the addition of distance to roads and trails. During final analysis of the summer horse habitat selection, the leading model was "thermal+habitat," explaining 17.2% of horse distribution (Table 4), and included the same variables as the winter model. The most appropriate fall model was the "thermal+habitat+disturbance" combination, explaining a relatively low amount of variance at 13.3% (Table 4). This model had the same variables as the spring model, with the addition of distance to seismic lines.

A similar type of relationship existed for thermal and habitat variables regardless of season. Habitat selection was positively related to distance to conifer and mixedwood forests and solar radiation (Table 5). During spring and fall, habitat selection increased with distance to roads/trails and seismic lines (Table 5).

Table 4. Final summary results depicting comparative model strength of combined themes of feral horse observations from global positioning system telemetry data collected during winter (1 November–31 March), spring (1 April–15 May), summer (16 May–15 September), and fall (16 September–31 October) of 2009 and 2010, and various landscape attributes. Bolded and italicized model indicates final model selection.

Theme	Component (Final spring analysis)[1]	K[2]	R^2 Winter	Spring	Summer	Fall
Null		1	0.00[3]	0.00	0.00	0.00
Thermal	D. conifer + D. mixedwood + GSR[4]	4	14.44	19.30	13.62	10.03
Thermal + Vegetation Type	D. conifer + D. mixedwood + GSR + ANS	6	***21.25***	30.28	***17.15***	11.47
Thermal + Veg Type + Water and Access	D. conifer + D. mixedwood + GSR + ANS + TRI	8	22.02	30.29	17.61	11.79
Thermal + Veg Type + Disturbance	D. conifer + D. mixedwood + GSR + ANS + D. roads/trails[5]	8	21.74	***31.48***	17.76	***13.26***
Thermal + Veg Type + Water and Access + Disturbance	D. conifer + D. mixedwood + GSR + ANS + TRI + D. roads/trails	9	22.70	31.56	18.45	14.12

[1]Component terms are defined in Table 1.
[2]Indicates the number of parameters used.
[3]McFadden's pseudo R^2 goodness of fit measure.
[4]ANS indicates avoided, neutral, or selected; TRI, topographic ruggedness index; GSR, global solar radiation.
[5]For the fall, disturbance consisted of D. roads/trails and D. seismic lines.

Table 5. Ranked influence of different variables in the leading resource selection function models by season of use for feral horses in the Alberta foothills. Data based on observations collected between October 2008 and October 2010.

Variable	$\beta^{1,2}$	SE^3
Winter (1 November–31 March)		
D. conifer	0.380	0.001
D. mixedwood	0.076	0.021
GSR[4]	0.200	0.000
Selected	0.580	0.031
Avoided	−1.140	0.033
Spring (1 April–15 May)		
D. conifer	0.250	0.002
D. mixedwood forest	0.088	0.032
GSR[4]	0.340	0.000
Selected	0.980	0.045
Avoided	−0.62	0.054
D. roads and trails	0.067	0.004
Summer (16 May–15 September)		
D. conifer	0.560	0.001
D. mixedwood forest	0.077	0.022
GSR[4]	0.140	0.000
Selected	1.030	0.035
Avoided	−0.310	0.031
Fall (16 September–31 October)		
D. conifer	0.450	0.031
D. mixedwood	0.074	0.002
GSR[4]	0.110	0.00
Selected	0.850	0.066
Avoided	−0.560	0.430
D. roads and trails	0.031	0.0033
D. seismic lines	0.110	0.006

[1]Beta coefficient.
[2]All β coefficients shown have a significance of $P < 0.0001$.
[3]Standard error.
[4]GSR values are $\times 10^{-3}$; GSR, global solar radiation.

Final RSF maps created for each season (Fig. 3) reflected the likelihood of habitat selection by horses based on different aggregate habitat conditions (vegetation types, terrain, and distances to disturbances) across the study area. Seasonal RSF maps were scaled in ArcGIS 9.3 to seven ordinal ranked categories of selection using quantile binning. Seasonal habitat suitability maps indicated that 14% of the landscape was selected, 42% of the landscape was avoided, while the remainder of the study area was neutral (i.e., used according to availability).

DISCUSSION

Home Ranges of Feral Horses

Previous work has shown that horse home ranges can vary considerably in size (McCort 1984), consistent with our findings. The average home range of horses examined here was 48 km^2, which was 33 km^2 larger than that found by Salter and Hudson (1982) within a similar environment in west central Alberta. Interpretation of our home ranges should be tempered by the 95% kernel ranges we used and risk of correlation among horses, which is known to underestimate home range size (Peridotto-Baldivieso 2012). Should this be the case, however, sampled horses would have even larger home ranges, further differentiating them from Salter and Hudson (1982). Although the larger home ranges in the current study could arise because of a difference in resource availability or exposure to disturbances between study areas, differences in study methodology (i.e., use of GPS collars here) may also influence home range size. The ability of GPS collars to continuously track horse movement throughout the year would effectively maximize home ranges. In contrast, Salter and Hudson (1982) relied on field observations, which occurred under a limited sampling period and intensity, and may have underestimated home range size.

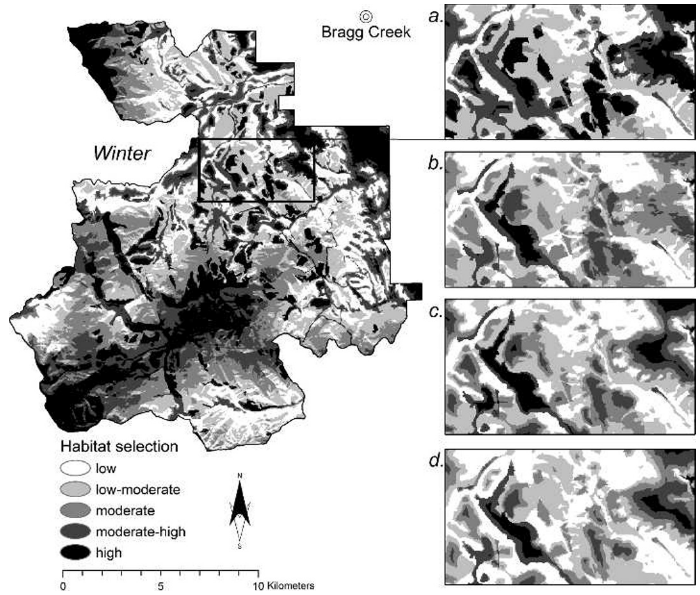

Figure 3. Maps depicting the likelihood of horse use for feral horses in the McLean Creek area of southwest Alberta, based on resource selection functions (RSFs) developed for the region. Subset maps represent the **a)** winter, **b)** spring, **c)** summer, and **d)** fall seasons.

The relatively stable home ranges across consecutive seasons suggested that these animals had territorial and home range fidelity, similar to the findings of Ganskopp and Vavra (1986). As feral horses are gregarious animals (McCort 1984), it is likely that home ranges mapped in the current study are representative of harems rather than individual animals. Although some horses appeared to use habitats at a greater intensity than others based on our data (i.e., horse 2, which had a very small home range), this was not supported by the strong association between initial harem sizes and home ranges. High variation in landscape diversity also ensured that each horse had access to all habitats, even within a relatively small area. Moreover, horse 2 occupied the most isolated (and least accessible) region, which may have led to a reduction in human disturbance. In contrast, harems situated closer to increased human activity (i.e., near public campgrounds) had larger home ranges. Larger home ranges in these areas could arise as horses move about to avoid interactions with humans (Laliberte and Ripple 2004), a finding supported by the RSF models from spring and fall when disturbances were relatively more important (Table 5). However, the large home range size of the collared horse nearest the campground (horse 4 in Fig. 1) may also have occurred because resources were more limited in this high traffic area. This region had the smallest proportion of (preferred) grasslands and shrublands of all home ranges.

Seasonal Selection by Horses

Distinct seasonal trends in habitat selection were observed, particularly for vegetation types. During summer, horses strongly selected for grasslands and riparian shrublands. Both these habitats have favorable herbage production (ASRD 2005), as well as the grasses and sedges specifically sought out by horses when foraging (Salter and Hudson 1979). Preferred species during summer and commonly found in grasslands and shrublands included *Deschampsia caespitosa, Festuca* spp., *Poa* spp., *Carex* spp., and *Phleum pratense*. Although depletion of forage could arise at this time of year given that cattle are using similar vegetation types as horses (Girard et al. 2013) and have similar diets to horses (McInnis and Vavra 1987), interspecific competition is unlikely during this time given the rapid growth and biomass increases observed, with maximum production values for grasslands ranging from 3 600kg $\cdot$ ha^{-1} to 4 000 kg $\cdot$ ha^{-1} in this region (ASRD 2005; Girard et al. 2013).

During fall, horses selected grasslands, but at a lower level than during summer, and avoided conifer forest, with all other vegetation types used according to availability. Reduced selection for grasslands during fall may be due to progressive depletion of available forage in habitats selected during summer by the combined grazing pressure from feral horses and domestic cattle (Girard et al. 2013). This in turn may account for the increased habitat selection for cutblocks during fall and winter, particularly given that horses are known to prefer high biomass areas (Fleurance et al. 2009).

Increased selection by horses for conifer cutblocks during winter contradicts Irving (2001) who found horses in the Upper Foothills of Alberta (350 km NW of this study) selected disturbed areas (e.g., roadsides, pipelines, and other developed lands) over pine cutblocks. The increase in selection for conifer cutblocks found here may be a strategy by horses to widen their search for remaining forage (Salter and Hudson 1979), particularly with depletion of forage within their primary grassland ranges. Similar to feral horses in the current study, cattle in Alberta avoided conifer cutblocks during summer (Kaufmann 2011). In combination, these results suggest forage in conifer cutblocks is less likely to be as depleted as other habitats (grasslands and riparian shrublands) by early winter. Finally, harvested conifer cutblocks that occur above the valley bottom are less susceptible to cold air drainage during winter (Henson 1952), and therefore have warmer conditions compared with valley bottom grasslands. Ambient temperatures from the GPS collars support this as mean temperatures during January were 4°C greater for horses occupying conifer cutblocks than those in lowland grasslands.

Increased selection for shrublands during spring coincides with the increased presence of shrubs in the spring diets of horses observed (based on fecal assessment) by Salter and Hudson (1979). Increased use of shrublands may arise because of a greater ability by horses to access these areas as snow melts, coupled with taller shrubs representing some of the only forage available after winter and prior to spring green up. This notion is also supported by the observation that the greatest aversion to conifer forests was evident during spring wherein snow is likely to persist. Overall, our findings suggest feral horses may be adapting seasonally to utilize what forage is available, accessible, and of suitable quality within their home ranges. This includes shifts throughout the year in the identity of primary habitat (i.e., vegetation) types.

Mechanisms Regulating Habitat Selection by Horses

Although habitat selection by feral horses differed by season, several common trends were evident. For all seasons, thermal aspects, in addition to core vegetation type, were important predictors of selection. Feral horses selected open areas away from conifer and mixedwood forests. Although forests may be used for temperature regulation by providing shade in summer and relief from wind and cold during winter (Musterud and Østbye 1999), our results indicated horses were not utilizing forest cover as expected. Instead, selection for the combined factors of solar radiation and greater distance from forested areas suggests horses may have been maximizing sun exposure, which would aid in winter thermoregulation. Similar observations have been made with cattle in Montana during winter (Keren and Olson 2007). Conversely, sun exposure may not have been high enough for horses to seek thermal cover during summer, and relatively cool summer temperatures in this environment (generally $<$ 30°C) may have limited the need for horses to seek shade. Forests also contain relatively low amounts of forage (Girard et al. 2013), which may dissuade horses from using these areas, at least when foraging. Finally, forests may be associated with greater exposure to predation. Horses are thought to be susceptible to predation, particularly from cougars (*Puma concolor*; Knopff 2010), and avoidance of forests may be an adaptive strategy to minimize this risk. Despite this risk, comparison of habitat selection in cutblock core and perimeter areas revealed horses did not exhibit differential use between these zones (data not shown).

Aversion by horses to roads, trails, and seismic lines may occur because of the large amount of human activity on and near these features (Laliberte and Ripple 2004). Roads and trails are traveled extensively by recreationalists, including hikers, cyclists, dirt bikers, off-highway vehicle riders, snowmobilers, and horseback riders. While this aversion was expected to be more prevalent in summer (i.e., during peak recreation use) than fall or spring, the opposite pattern was observed. Horses may be avoiding linear features during the transitional seasons due to a reduction in concealment cover. Areas adjacent to trails are where the majority of deciduous woody species (shrubs and trees) are found, and spring and fall would coincide with periods prior to leaf-out and after leaf-fall, respectively. Although we hypothesized that horses could be using linear features as movement corridors, this did not occur. Horses may also avoid linear features because the latter can attract predators (Whittington et al. 2005). Caution should be exercised in interpreting horse selection patterns during the short, transitional spring and fall seasons, as a smaller sample size of observations could result in less robust RSF models, and more variability may be expected in horse use within these seasons from year to year.

Water and topography did not affect habitat selection by horses, regardless of season. The lack of a water association corroborates Salter and Hudson (1979) who concluded that water was not limiting for horses in the Alberta foothills. Moreover, the finding that ruggedness was not a factor influencing habitat selection suggests topography (i.e., elevation, slope, and aspect) does not pose the same limitation for horses as it does for cattle (Kauffman 2011).

Across all seasons, observed RSF models accounted for moderate variation in horse distribution (13.3–31.5%), and could indicate that other explanatory factors were not captured in our assessment of habitat selection. Model fit was greatest during spring, which was unexpected because spring is one of the shorter and more variable seasons. However, rapidly changing conditions at that time (i.e., coincident with snowmelt and green-up) may have led to more predictable behavior by horses as they attempt to maximize recovery following winter. In contrast, the lowest model fit was during fall, consistent with the notion that this transitional season can bring widely varying foraging conditions depending on the previous summer's growth coupled with variability in the onset of senescence. Finally, we acknowledge the potential limitations imposed by low sample sizes (number of horses) and any interactions among harems across the study area in explaining feral horse use during the 2-yr study period.

MANAGEMENT IMPLICATIONS

Overall, our results indicate that in southwest Alberta, relatively small amounts of the landscape are preferentially selected by horses, particularly grasslands and shrublands across all seasons, and during winter, harvested conifer forests (cutblocks). In addition to vegetation type, selection by feral horses was influenced by other habitat characteristics, primarily distance to forests and sun exposure. Although horses used all areas of the landscape, selected grassland habitats had the smallest footprint and are likely the most sensitive to human disturbance as horses avoid roads and trails travelled by people. Future increases in recreational activity may continue to shift feral horse selection from conventional primary range (grasslands and shrublands) into alternate habitats, with any displacement posing a threat to horse survival and localized range health. Future monitoring programs to track recreational use may be useful to determine how these changes alter habitat selection by horses. Moreover, this process may be further complicated by ongoing grassland declines due to shrub encroachment (Burkinshaw and Bork 2009). RSFs generated in this study should enable land managers to map existing and additional primary habitats likely to be used by horses, as well as establish seasonal carrying capacities based on temporal changes in horse use. For example, as horses demonstrated the narrowest selectivity for specific habitats during winter, this period could pose the greatest limitation to horse survival. Consequently, winter habitats may be used to establish yearlong carrying capacities of feral horses in the region.

Selection of harvested conifer forests in winter could also be problematic and lead to heightened land use conflict between the forest industry and feral horse management. For example, it is unknown whether, and if so how, increased horse use of conifer cutblocks may change tree seedling damage and regeneration. Similarly, it is unknown whether horse use of cutblocks during winter is influenced by existing levels of grazing from horses, cattle (i.e., during the previous summer), or both, within adjacent primary habitats, or other conditions. High accumulated use of grasslands due to combined horse and cattle grazing (Salter and Hudson 1980; Girard et al. 2013) increases the likelihood of changes to horse behavior. Further study is needed to determine the impact and mechanisms regulating seasonal horse grazing in cutblocks of the region, particularly in conjunction with other land uses.

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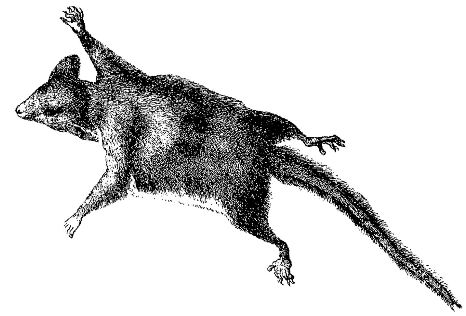

Commentary

Management implications of the ecology of free-roaming horses in semi-arid ecosystems of the western United States

Erik Beever

Abstract Compared to other ungulates of North America, free-roaming horses (*Equus caballus*) possess a unique evolutionary history that has given rise to a distinct suite of behavioral, morphological, and physiological traits. Because of their unique combination of cecal digestion, an elongate head with flexible lips, and non-uniform use of the landscape, horses represent a unique disturbance agent in semi-arid ecosystems of the western United States. Consequently, it is inappropriate to assume that influences of horses on the structure, composition, function, and pattern of arid and semi-arid ecosystems will mirror influences of cattle or other artiodactyls. Although management areas for free-roaming horses occupy 18.6 million ha of land across western North America, we know relatively little about how western ecosystems and their components have responded to this uniquely managed ungulate. I draw on my research of horse habitats in the western Great Basin (U.S.A.) to examine predictions of horses' unique influence, and advocate for continued research to refine our understanding of synecological relationships among horses and diverse ecosystem components in arid and semi-arid regions.

Key words *Bos taurus*, *Equus caballus*, life-history traits, management, semi-arid ecosystems

Management of free-roaming horses (*Equus caballus*) in western North America has proven to be an ongoing political controversy in the 20th and 21st centuries, and the management challenge has escalated in importance since passage of the Wild Free-Roaming Horse and Burro Act (1971). Local, state, and federal politicians, often using incomplete or faulty knowledge, have increasingly tried to constrain the outcomes of wild horse and burro (*E. asinus*) management (Zarn et al. 1977, Wagner 1983, Bellisle 1997, Bama 1998). Galvanization of interest groups (such as ranchers, animal-rights activists, hunters, conservationists, and horse advocates) is increasingly forcing managers to adopt more rigorous, scientifically based methods and analyses to justify management actions. Although free-roaming horses have inhabited western North America since the end of the 16th century, little synecological research has been done to quantitatively characterize how they interact with ecosystem components there. The National Academy of Sciences recognized the importance of potential effects of free-roaming horses on native biota when they commissioned 6 studies in 1980 to address those effects. However, the database for formulating a sound management program was very limited (Wagner 1983).

This article is written in response to the perceptions of some managers, biologists, and the general public that free-roaming horses are ecologically comparable to domestic cattle (*Bos taurus*) (e.g., equivalencies used in management calculations of Animal Unit Months [AUMs]) and refill niches occupied by equids prior to the Pleistocene megafaunal extinctions (e.g., Zarn et al. 1977, Wagner 1983). Because of the former perception, their influence on structure, composition, and function of semi-arid ecosystems is explicitly or (more often) implicitly assumed to parallel cattle influences. I describe the biology of free-roaming horses in western North America, and draw from my research in 9 herd management areas of the western Great Basin (U.S.A.) to highlight ways that horses constitute a unique influence on western landscapes. To further clarify the role that horses play in disturbance regimes of western North America, I conclude that further synecological research is needed to characterize how this herbivore interacts with other ecosystem components.

History and genetics of wild horses in North America

Although equids arose and diversified in North America (Simpson 1951), they were one of 18 taxa known from the Great Basin to vanish from the continent at the end of the Pleistocene around 10,000-14,000 years ago (Grayson 1993). This wave of extinctions resulted from hunting by prehistoric humans, climate change (and resulting vegetation shifts), or a combination of the two (Martin 1984, references in Martin and Klein 1984, Grayson 1993). After equids were absent from North America for at least 10,000 years, domestic horses were brought to the southwestern United States by Spaniards near the end of the 16th century (Zarn et al. 1977, Wagner 1983, Fradkin 1989). Reductions in predator numbers, increased availability of water due to construction of "wildlife guzzlers," increased mobility facilitated by fences in disrepair, and the longstanding commensal relationship between horses and humans resulted in the number of free-roaming horses in the United States rising to an estimated peak of 2-7 million animals during the 19th century (Ryden 1978, Thomas 1979). However, numbers of horses declined steadily during the late 19th through the mid-20th century because of persecution, domestication, and other means of removal, facilitated by the Taylor Grazing Act of 1934 (Wagner 1983).

With protection afforded by the Wild Free-Roaming Horse and Burro Act of 1971, which was enacted largely in response to inhumane treatment of free-roaming horses, numbers of horses on public lands rose sharply from approximately 17,300 in 1971 to an estimated peak of 57,200 in 1978 (Wagner 1983, Anonymous 1997). Given the changes in selective forces mentioned above, many herds have exhibited annual population growth rates of 20% and higher (Wolfe 1980, Eberhardt et al. 1982, Anonymous 1997). According to a recent Bureau of Land Management (BLM) estimate (30 May 2002), free-roaming horses in the western United States currently number around 38,815 individuals on 208 herd management areas encompassing 17.5 million ha of BLM-administered lands across 10 states. In addition, lands administered by the United States Forest Service (that contain other jurisdictions) contribute an additional 1.1 million ha and approximately 1,600 animals (Anonymous 1997). In Nevada, the state with the most animals and herd areas, nearly 30% of the state's area is occupied by horse and burro herd management areas (Anonymous 1997, Hammond World Atlas Corporation 2000).

Several authors have suggested that current populations of free-roaming horses in the western United States are not the result of a single introduction 350-450 years ago but rather the amalgamation of repeated introductions of domestic horses and subsequent mixing with already established feral herds (e.g., Ryden 1978, Wagner 1983). Similarly, Beebe and Johnson (1964) suggested that due to repeated interbreeding with released or abandoned domestic horses, the free-roaming horse of the western United States differs little from other small domestic horse breeds. Ryden (1978) believed that descendants of the original Andalusian horses brought from Spain to the Americas numbered in the hundreds and had largely been removed to captivity. Bowling (1994) compared 19 polymorphic loci from blood samples of 975 free-roaming horses from 7 Great Basin sites with samples from 16 domestic horse breeds and found no difference in either number of variants or expected heterozygote frequency. From pairwise comparisons of Nei's genetic distance measurements, Bowling (1994) concluded that Great Basin horses originated from Iberian, American saddle horse, and draft horse breeds.

Because free-roaming horses are relatively easy to study (in that they are large, easily detectable, diurnal, and found in groups), there is a relatively well-developed literature on various aspects of horse biology. These studies include descriptive studies of diet (e.g., Hansen 1976) and of partitioning of dietary and habitat niches when horses are sympatric with other native and domestic ungulates (e.g., Hubbard and Hansen 1976, Olsen and Hansen 1977, Hanley and Hanley 1982, Krysl et al. 1984, McInnis and Vavra 1987, Coates and Schemnitz 1994). Behavioral ecology research has suggested that because of their large body size and great speed and power, free-roaming horses are socially dominant when interacting with native ungulates of the Great Basin, producing effects on wildlife species different from those produced by cattle (Meeker 1979, Berger 1985).

Moreover, studies have addressed genetic questions related to herd uniqueness and concern for minimum viable populations (e.g., Bowling and Touchberry 1990, Goodloe et al. 1991, Gross 2000) as well as issues of social biology (Feist 1971, Miller 1980, Berger 1986, Turner et al. 1992) and reproductive biology (Eberhardt et al. 1982, Seal and Plotka 1983, Garrott et al. 1991). Immunocontraception has been deemed the most humane and socially acceptable method of population control, and studies have proliferated in recent years to fine-tune this technique for management (e.g., Turner et al. 1997, Powell and Monfort 2001).

Although our understanding of the autecology (i.e., the biology of organisms or populations, without regard to other elements in the ecosystem) of horses and their interactions with other ungulates has improved greatly in recent decades, understanding how free-roaming horses interact with the non-ungulate components of semi-arid ecosystems is only beginning (Crane et al. 1997, Fahnestock 1998, Fahnestock and Detling 1999, Peterson 1999). This is especially curious, given that research on grazing impacts on plant communities was given the highest priority by the Committee on Wild and Free-Roaming Horses and Burros over 2 decades ago (Wagner et al. 1980). Given the large extent of land that horse management areas occupy, an understanding of how horses affect various components of their environment seems vitally important. In studying 2 sagebrush-dominated elevational strata across 9 mountain ranges of the Great Basin during 2 wet years, I found that horse-occupied areas possessed more deer mice (*Peromyscus maniculatus*), more depauperate rodent guilds, fewer ant mounds, and more plant species, in addition to lower grass and shrub cover, than did horse-removed areas (Beever et al. 2003). This result suggests that there remains room to fine-tune both our understanding of free-roaming horses' roles in semi-arid landscapes and the determination of ecologically appropriate herd sizes.

Comparison of free-roaming horses and other large herbivores in western North America

Compared to other ungulates of North America, horses possess a unique evolutionary history that has given rise to a distinct suite of behavioral, morphological, and physiological traits (Simpson 1951, Hafez et al. 1969, Feist and McCullough 1975, Janis 1976, Berger 1986). In contrast to other large Intermountain West ungulates that are ruminants, horses are cecal digesters (Janis 1976, Hanley and Hanley 1982). Combined with their large body size, this type of digestion places more time–energy constraints on the animal, meaning that the free-roaming horse is one of the least-selective ungulate grazers across most of western North America (Hanley and Hanley 1982). Thus, fewer plant species may remain ungrazed in areas occupied by free-roaming horses compared to areas grazed by other ungulates. Although elk (*Cervus elaphus*) and feral burros can also consume a broad spectrum of food items, these species occupy only small portions of the area encompassed by the 18.6 million ha of wild-horse herd areas. This use of a lower-quality diet requires that horses consume 20–65% more forage than would a cow of equivalent body mass (Hanley 1982, Wagner 1983, Menard et al. 2002). In addition, horses possess a more elongate head and more flexible lips than cattle and, unlike cattle, have upper front incisors. Consequently, they can trim vegetation more closely to the ground than do cattle, sometimes delaying the recovery of plants (Symanski 1994, Menard et al. 2002).

Differences between free-roaming horses and cattle become more numerous and more pronounced as one scales up to investigate how the biology of horses translates into grazing consequences at the landscape scale (i.e., within and across mountain ranges). For example, horses often segregate elevationally from sympatric cattle, using steeper slopes and occupying higher elevations (Pellegrini 1971, Ganskopp and Vavra 1986). This difference may stem in part from the fact that free-roaming horses in North America are related to the truly wild Przewalski's horse (*E. c. przewalskii*) native to the cooler Asian and European steppes (Wagner 1983, van Dierendonck and Wallis de Vries 1996). Several authors have noted horses' disproportionately high use of ridgetops and high benches (Pellegrini 1971, Miller 1980, Keiper and Berger 1982, Ganskopp and Vavra 1987). I agree with Pellegrini's (1971) premise that such behavior may be an effort to maximize the viewshed of the horse, rather than pest avoidance as suggested by Keiper and Berger (1982). This behavior may represent an evolutionary vestige of a "flight" response to predation predominantly in the past, a phenomenon not generally observed in cattle on western landscapes.

Another difference between cattle and free-roaming horses is that horses tend to use semi-arid landscapes more heterogeneously at some spatial scales than do cattle. At the smallest scales, horses will use a few trails repeatedly to cross the landscape (Figures 1*a*, 1*b*), whereas cattle more often graze all portions of an area with similar intensity (Menard et al. 2002). Horses restrict themselves to fewer pathways partly because of their territoriality—they patrol the territory boundary of the group repeatedly (Pellegrini 1971, Zarn et al. 1977). Horses also use only a few trails to travel to and from water (particularly during the driest seasons), traveling farther from water each day than do cattle (Pellegrini 1971, Green and Green 1977). In contrast, cattle tend to stay close to springs or riparian areas throughout the day and season, unless managed otherwise (review in Kauffman and Krueger 1984). Cattle also create detectable trails in some

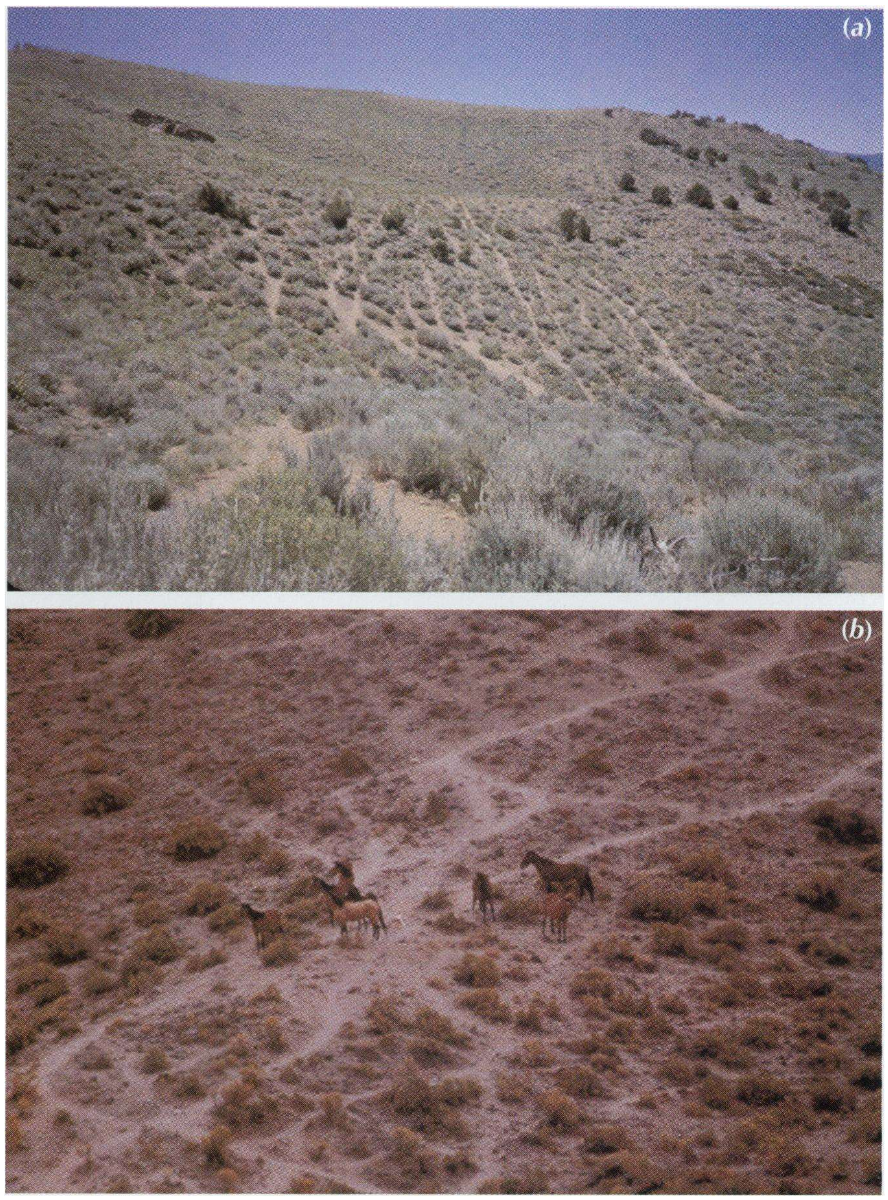

Figure 1. Examples of heterogeneity in habitat use by free-roaming horses in semi-arid ecosystems. (*a*) Trails primarily created by horses traveling across a sagebrush-dominated hillside on Dogskin Mountain, western Nevada. (*b*) Trails created by horses in salt-scrub habitat, central Nevada.

instances, but often the number, length, and spatial extent of cattle trails are less than horse trails (personal observation). As a means of territorial establishment and boundary marking (Pellegrini 1971), free-roaming horses concentrate their defecations in dung posts and stud piles that can reach over 60 cm in height and >10 m^2 in extent (Pellegrini 1971). In contrast, unless they are near a watering source, cattle distribute themselves and their defecations more uniformly across the landscape. At the landscape scale, concentration of cattle at watering areas constitutes an exception to the generality of greater heterogeneity in habitat use by horses compared to cattle. Heterogeneous use of landscapes by free-roaming horses should translate into different effects on the processes of soil stabilization, water retention, and nutrient cycling, as well as on vegetative characteristics, than mosaics created by cattle grazing.

To test this prediction, I examined the ecosystem consequences of heterogeneous use of a landscape by free-roaming horses in field research in western Nevada at sites in high- and low-elevation big sagebrush (*Artemisia tridentata*) habitats. At the broadest (landscape) scale, sites from which horses had been removed for 10-14 years exhibited significant differences in vegetative characteristics (e.g., grass cover, shrub cover, species richness) from horse-grazed sites (Beever 1999, Beever et al. 2003). In addition, I observed trends of the same magnitude and direction when I compared, within horse-grazed sites, randomly located line-intercept transects with transects placed alongside established horse trails (25-30 cm from the trail center) (Beever 1999). Specifically, randomly located transects at horse-occupied sites exhibited an average of 1.2-9.5 times greater grass cover, lower forb cover (at all but the lowest-elevation site), 1.6-2.5 times greater shrub cover, and greater species richness of shrubs than did "horse-trail" transects. This second-level (smaller-scale, within-site) comparison corroborates the existence of heterogeneity in grazing intensity at both the landscape and site scales.

In addition, I used a handheld penetrometer to measure penetration resistance of soil surface horizons (a surrogate of soil-surface hardness) at both horse-removed and horse-grazed sites. I sampled 10 points within a 1-m-diameter circle at each of 25 locations per 1.82-ha site (see Beever et al. 2003), and hypothesized that relative variability in hardness values among locations within a site would be higher at horse-grazed sites, due to horses' heterogeneous use of the landscape. Relative variability (as measured by CV) in penetration resistance averaged 1.65 times higher at horse-occupied sites in 1997 sampling, even though resistance was 3.0 and 4.5 times higher at horse-occupied sites at high elevations and low elevations, respectively (E. Beever, unpublished data; Beever et al. 2003). Sampling in 1998 showed an even greater difference in penetration resistance, as horse-occupied sites exhibited 17.4 times higher penetration resistance than horse-removed sites at low elevations and 2.9 times higher penetration resistance at high elevations (E. Beever, unpublished data; Beever et al. 2003). The much lower means dictated that horse-removed sites in 1998 did not have greater variability in soil hardness but rather exhibited significantly lower relative variability than horse-occupied sites. Thus, the greater amount of absolute variability observed at horse-occupied sites in 1998 was overridden by the vast difference in mean hardness.

Due to the differences noted above, it is inappropriate to assume that effects of grazing by free-roaming horses are similar in nature or in magnitude to effects of grazing observed for cattle or other species. Even among the closely related African equids, Klingel (1972) noted considerable differences in social organization. Differences such as grouping behavior can affect movement patterns in horses (Berger 1986) and other ungulate populations (Bailey 1984, Feh et al. 1994). Ways in which greater heterogeneity in habitat use by horses (relative to cattle) at several spatial scales translates into effects on patch dynamics, landscape ecology, and a cumulative measure of ecosystem function remain unknown. Consequently, in light of the tremendous amount of controversy engendered by issues of horse management (Thomas 1979, Wagner 1983, Linklater et al. 2002), it is important to study both the interactions of free-roaming horses with other species and their general effects on ecosystems. Recent studies of horses in more mesic ecosystems (e.g., Levin et al. 2002, Menard et al. 2002) have provided an insightful first step in that direction. For example, in 2 wetlands in France, Menard et al. (2002) found that cattle used forbs and shrubs much more than did horses, whereas horses spent more time feeding in short-grass areas and maintained a mosaic of patches of short and tall grass. However, these ecosystems have horses present in only limited numbers and over a limited area, and their response to horse grazing is likely fundamentally different than what would be expected in semi-arid ecosystems where thresholds and nonlinear dynamics dominate (Mack and Thompson 1982, Laycock 1991).

Just as it is questionable to extrapolate past research on grazing or browsing effects of other ungulates to horses, extrapolating results of numerous published studies on domestic horses to free-roaming populations may be problematic, particularly with respect to behavior. Since the time horses were first released into North America ca. 350-450 years ago, they may have diverged from their mixed domestic ancestry in some traits. When compared to the average generation time of free-roaming horses, approximately 3-6 years, 350-450 years translates into approximately 75-150 generations during which natural selection could have acted. However, because free-roaming herds have continued to receive immigrants from and interbred with domestic horses, differences between domestic and free-roaming animals more likely simply reflect phenotypic plasticity in horses experiencing different availability of forage, demographics, and available habitats, rather than genetic divergence. For example, eliminative behavior in free-roaming horses is often concentrated at "stud piles" along edges of territories dominated by harem-possessing stallions (Pellegrini 1971, Miller 1980). In contrast, such concentration of feces is generally not observed to the same degree in domestic horses, even in large pasture areas (Ödberg and Francis-Smith 1977). Because eliminative behavior may direct horse movements and affect nutrient cycling at various scales, spatial distribution of grazing effects may thus differ between domestic and free-roaming horses.

Another behavior that may differ between domestic and free-roaming horses is grouping. Although domestic horses may exhibit social stratification (Hafez et al. 1969), they do not appear to exhibit the full complement of associations (e.g., harem bands, multiple male and female bands, and bachelor groups) observed in free-roaming horses (Feist 1971, Berger 1977, Zarn et al. 1977, Miller 1980). However, in domestic settings the proportions of horses that are mares, yearlings, geldings, and studs will greatly influence the diversity of associations observed. Thus, knowledge of the grazing ecology of domestic horses in captive settings (Hafez et al. 1969, Reiner and Urness 1982) may not accurately predict expected consequences of free-roaming horse grazing, due to differences in their behavior. Free-roaming horses are believed by some to possess harder hooves and have ability to last for longer periods without water than some domestic horse breeds (S. Kipping, BLM, Washington D.C., personal communication), though this contention has not been demonstrated with empirical data.

In summary, because of the behavioral differences between captive and free-roaming horses (which may or may not reflect heritable traits), we cannot uncritically rely on studies of domestic horses to help us understand ecosystem response to free-roaming horses. More investigations are needed on free-roaming horse populations to predict how they will influence their surroundings.

Management considerations and conclusion

In addition to the differences noted above, a final suite of differences between horse and cattle influences on semi-arid landscapes arises from the unique management status of horses. Free-roaming horses are not managed as wild or as domestic animals; they currently occupy a unique political status among large mammals of North America. Although cattle and free-roaming horses are of similar size, cattle generally are managed more intensively (e.g., with fencing exclosures and enclosures, rotation grazing, herding, provision of salt licks and supplemental water, etc.) than are horses. In contrast, horses by law must be managed under a "minimal management strategy." For example, other than during periodic removals, many free-roaming herds of horses are not fenced. In contrast to other wild ungulates, however, hunting of horses is not permitted, as mandated by the Wild Free-Roaming Horse and Burro Act of 1971. These policies constrain possible management strategies and mean that distribution of horse grazing across semi-arid landscapes will diverge greatly from cattle distribution.

Herbivory and trampling may occur across a larger percentage of the physiographically heterogeneous Intermountain West with the addition of wild horses (*sensu* Symanski 1994), as in Australia (Symanski 1994). Other ungulates such as mule deer (*Odocoileus hemionus*) and bighorn sheep (*Ovis canadensis*) also use upland and steep areas (Ganskopp and Vavra 1987). These ungulates are substantially lighter and possess smaller hoof-surface areas than horses (Symanski 1994); consequently, native ungulates may exert less physical impact on plants and upper soil horizons than do free-roaming horses at similar densities.

My purpose in this article has not been to argue whether horses in western North America should be considered "wild" or "feral." I refer readers to the thoughtful and well-balanced treatment of that controversy by the Committee on Wild and Free-Roaming Horses and Burros (Wagner et al. 1982). My purpose has been to highlight the uniqueness of the free-roaming horse among large herbivores of western North America, and to explore ways in which their differences from other ungulates may translate onto managed landscapes. Paleontological and other lines of evidence have suggested that large-bodied grazers existed at low densities across the Intermountain West from the Pleistocene extinctions of 10,000–14,000 years ago until >180 years ago (Mack and Thompson 1982; Milchunas et al. 1988, Grayson 1993). Relative to grassland and savannah ecosystems, ecosystems that have experienced herbivory infrequently or at low intensity over evolutionary time may be less resilient to contemporary nonnative grazing, and may require more careful monitoring of responses to grazing of both plants and other ecosystem components to avoid deterioration.

Determining whether extant free-roaming horse populations exhibit density-dependent population regulation is complicated by at least three factors, reducing the utility of this consideration in management decisions: 1) horse population levels are likely kept below the carrying capacity of landscapes by periodic removals; 2) ability of management areas to sustain horses depends on the degree of niche overlap between horses and native herbivores in the area (Wagner 1983); and 3) density-independent mortality (e.g., from catastrophes or extremely severe conditions) occurs periodically.

In closing, I acknowledge that results from ecological research on free-roaming horses will be implemented in the context of a highly complex sociopolitical arena (Wagner 1983, Boyles 1986, Linklater et al. 2002). Given current legislation, there is no question that some number of free-roaming horses will be maintained in the United States. The primary management questions are how many horses there should be, how they should be distributed, and, following these, how to control horse numbers (Wagner 1983). With a clearer understanding of horses' effects, appropriate management levels can be adapted to facilitate grazing intensities and spatial mosaics that explicitly consider all species occupying public lands.

When mandates from ≥2 types of legislation (e.g., Wild Free-Roaming Horse and Burro Act of 1971, Endangered Species Act) come into conflict, I recommend the use of directed ecological research, open communication, and varied means of compromise, achieved through education and consensus-building. Under this strategy, well-planned synecological research and monitoring may be used to direct and bound options presented to the general public. For example, public concern regarding competition between free-roaming horses and either cattle or browsing mule deer or bighorn sheep is widespread. However, conditions necessary to demonstrate ecological competition (i.e., mutually reduced fitness) are difficult to achieve in the field (Wagner 1983), and the contention could be satisfactorily assessed only through replicated factorial experiments with AUMs kept constant across treatments within large-scale enclosures. Numerous aspects of horse management have been driven by political and social forces without sufficient biological understanding (Beever and Brussard 2000, Linklater et al. 2002). Therefore, I recommend enlightening societal concerns with adequate relevant ecological data when determining appropriate management levels (Boyles 1986).

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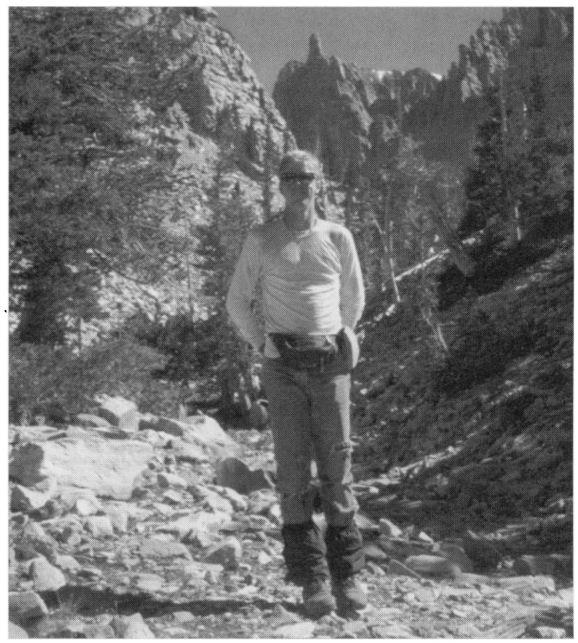

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BACKGROUND FOR NEPA REVIEWERS:

GRAZING ON FEDERAL LANDS

February 1994

U.S. Environmental Protection Agency
Office of Federal Activities
401 M Street SW
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BACKGROUND FOR NEPA REVIEWERS - GRAZING ON FEDERAL LANDS
INTRODUCTION

The primary purpose of the Guidance for NEPA Reviewers - Grazing On Federal Lands is to assist U.S. Environmental Protection Agency (EPA) staff in providing scoping comments and comments on National Environmental Policy Act (NEPA) documents associated with grazing on Federal lands, such as grazing Environmental Impact Statements (EISs) and Resource Management Plans. Pursuant to NEPA and Section 309 of the Clean Air Act (CAA), EPA reviews and comments on proposed major Federal agency actions significantly affecting the quality of the human environment. This document has been developed to assist the EPA reviewer in considering issues related to grazing in the development of NEPA/Section 309 comments.

This guidance is not intended to be all inclusive; rather, the document focuses on EPA's major concerns with surface and ground water, soils, and ecosystems as related to livestock overgrazing and provides technical background material explaining these issues. It does not restate traditional NEPA concerns about impacts on archaeological resources, economics, and so on, but rather addresses the technical environmental concerns related to overgrazing.

EPA realizes that rangeland management is often complex, and recognizes that each livestock grazing operation and each EIS is unique. Thus, reviewers will have to conduct additional analyses to fully understand projected impacts. The reviewer should not rely solely on this document as a definitive list of potential impacts or areas that should be covered by NEPA documentation. This document is more of a guide or introduction to issues associated with livestock overgrazing on Federal lands and does not replace early involvement in the NEPA process, defining objectives, developing alternatives, and determining effects based on knowledge of the issues and characteristics of specific areas.

Overview of Grazing Practices and Associated Impacts

Grazing on the open ranges of the Great Basin began in the mid 1800's and became a major industry in the western U.S. as early as the 1870's, with peak numbers of cattle and sheep being grazed by 1890. By 1900, many unrestricted lands were overstocked and significantly, sometimes even permanently, impacted. Impacts included trampled and compacted soils, lowered water tables in some areas, and replacement of quality vegetation with less desirable, more shallow-rooted species. As early as 1889, writers acknowledged that destructive grazing appeared responsible for denuding slopes of vegetation, increased runoff, erosion, and severe flooding in some western States (Gifford, NRC 1984).

In 1934, the system of free access to Federal lands ended with the passage of the Taylor Grazing Act and the establishment of the Division of Grazing, later to become the Bureau of Land Management, within the Department of the Interior. Although the Act was intended to rehabilitate rangelands, livestock numbers were not controlled and little rehabilitation occurred. This act was the first of many statutes directing the use of public lands for grazing. These statutes include the Multiple Use - Sustained Yield Act of 1960, the Forest and Rangelands Renewable Resources Planning Act of 1974, the National Forest Management Act of 1976, the Federal Land Policy and Management Act of 1976, and the Public Rangelands Improvement Act of 1978. National grasslands were bought under Forest Service management through the Bankhead-Jones Farm Tenant Act. The Fish and Wildlife Service oversees grazing on National Wildlife Refuges and in National Parks.

Both the Bureau of Land Management (BLM) and the Forest Service, acting as caretakers for lands under their jurisdiction, use an allotment system to control livestock grazing on Federal lands. Ten year renewable permits are issued for each allotment with the total fee based on the number of livestock and length of stay, calculated in terms of Head Months (HMs), or Animal Unit Months (AUMs). The Forest Service defines a Head Month as one month's use and occupancy of the range by one animal (one weaned or adult cow with or without calf, bull steer, heifer, horse, burro, mule or 5 sheep or goats). An AUM is defined as the amount of forage needed to support a 1000 pound cow and calf or 5 sheep for one month and consists of between 800 to 1000 pounds of forage. Currently, Federal grazing allotments cover approximately 30 percent of the total 853 million acres grazed nationwide, with most grazing on Federal Lands occurring in the western U.S.

Both the Forest Service and the BLM have separate requirements that apply to grazing. As part of their management responsibilities, both the Bureau of Land Management and the Forest Service develop area-specific management plans called Resource Management Plans or Forest Plans. These plans provide a comprehensive framework for managing and allocating uses of public lands and resources, such as fluid and locatable minerals, riparian resources, wildlife and fish habitat, and livestock grazing. Based on the management plans, the Bureau of Land Management and the Forest Service develop allotment management plans and issue grazing permits for those allotments, which present decisions on grazing at a more detailed level. More detail on these activities is provided in Forest Service and BLM Handbooks.

Each of these activities or decisions, ranging from developing a plan to issuing a lease or taking a specific range management action, may be subject to NEPA review. Typically the Bureau of Land Management or the Forest Service prepares an EIS for each Resource Management Plan or Forest Plan. For more detailed or allotment-specific activities, additional NEPA documentation is usually tiered (based on the existing Resource Management or Forest Plan EISs). Activities that are not addressed in existing NEPA documentation may require additional NEPA review, such as an Environmental Assessment (EA) and/or an EIS, if the proposed action "significantly affects the quality of the human environment." Under the CAA Section 309, EPA has the authority to review and comment on each EIS.

Despite attempts to control environmental impacts caused by overgrazing and recent improvement in rangelands according to some sources (Platts, 1990), many problems still exist in both upland and riparian areas. Issues characterizing upland areas, especially in arid environments, include the sensitivity of desert ecosystems and the extreme difficulty in reclaiming upland areas after impacts have occurred. Riparian areas are often of more concern to the public and Federal land managers for several reasons. Cattle tend to congregate in riparian areas, using them for shade and drinking water and spending a disproportionate amount of time foraging and trampling these areas rather than upland areas, posing a potentially higher level of damage. Also, riparian areas support a higher diversity of terrestrial and aquatic organisms than upland areas and provide critical habitat for both terrestrial and aquatic organisms. Erosion caused by overgrazing can reduce a streambank's water retention capabilities, lowering the surrounding water table and often changing the character of the stream from perennial to intermittent (GAO, June 1988a). Livestock and wildlife overgrazing can cause direct impacts on upland and riparian areas, such as loss of vegetation and soil compaction that lead to indirect impacts on the hydrology of an area and the ecosystems, both terrestrial and aquatic, that rely on it.

The remainder of this document describes important issues associated with the grazing of livestock on Federal Lands. Specifically, the document is arranged in the following sections:

- technical description of grazing;
- potential environmental impacts, both direct and indirect, associated with grazing;
- possible prevention/mitigation measures;
- types of questions that can be posed as part of the Agency's response to review of NEPA documentation; and
- explanation of the statutory and regulatory framework under which grazing on Federal lands occurs.

As discussed above, this document does not substitute for indepth knowledge of rangeland management concepts and site-specific issues.

TECHNICAL DESCRIPTION OF GRAZING ON FEDERAL LANDS

National and Regional Perspectives

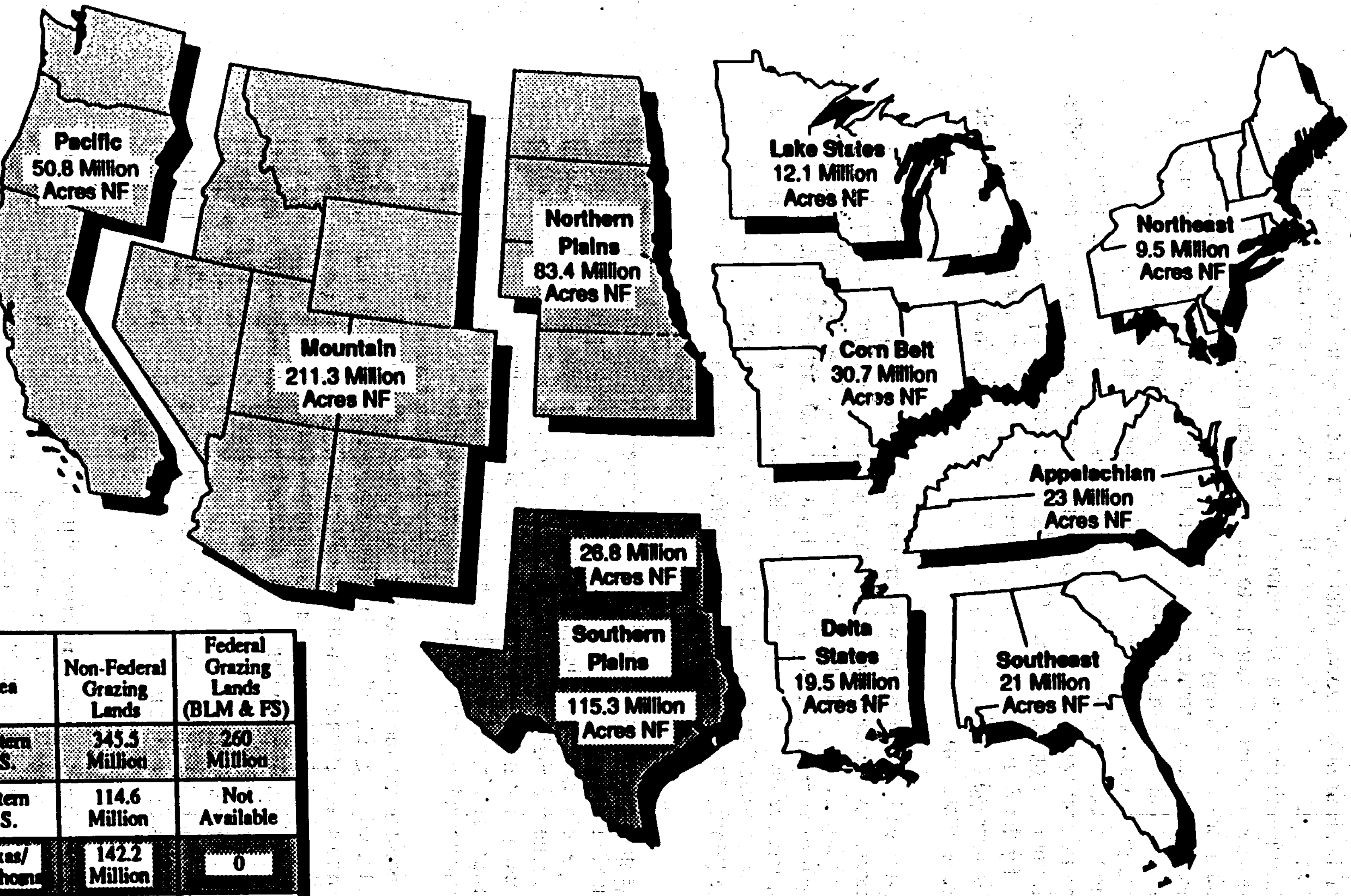
Over 95 percent of livestock grazing on Federal lands occurs in the western U.S. The BLM and the Forest Service manage a total of 461 million acres of public land. Of this, approximately 367 million acres are in the western U.S.[1] with grazing allotments covering about 70 percent of this area. Specifically, the BLM has approximately 165 million acres with approximately 22,000 separate grazing allotments (BLM, 1990). Of the Forest Service's 191 million acres, 104 million acres are allotted to grazing (95 percent of these allotments are located in the west) with approximately 50 million acres classified as suitable for grazing (e.g., slopes are not too steep) (GAO, May 1991). This compares with private grazing lands of approximately 603 million acres nationwide with 372 million acres of private grazing acreage in the western states[1]. Figure 1 shows both Federal and non-Federal grazing lands in the U.S. Texas has the most non-federal grazing lands with approximately 115 million acres; however, there are no BLM or Forest Service lands in Texas (Department of Agriculture, 1982).

BLM and the Forest Service manage public lands through allotments that typically have ten year permits and sometimes yearly or seasonal licenses (which are more specific than 10 year permits). Permits specify the number and type of livestock, an authorized season of use, and the AUMs (a measure of the amount of grazing available). The acreage required to provide one AUM varies from region to region, ranging from a low of 6.1 acres in Montana to a high of 21.8 acres in Nevada. The overall average AUM is 13.7 acres. The average grazing allotment is approximately 8,500 acres (13 square miles) with allotments as small as 40 acres and as great as 1 million acres (GAO, June 1988b). In many cases, allotments are interspersed with private lands, creating the checkerboard pattern seen on most Federal lands maps. This checkerboard pattern hampers effective control by Federal land managers, and requires constant cooperation between land mangers and ranchers.

According to 1990 statistics, BLM had about 165 million acres of grazing allotments, with almost 20,000 operators and 4 million head of livestock using 13.5 million AUMs (BLM, 1990). In 1986, the Forest Service had about 102 million acres in grazing allotments (in 36 states) with 13,805 permits using a total of 8.6 million AUMs. GAO estimates that 25 to 30 percent of the Forest Service allotments are in a declining condition and/or are overstocked.

As described above, Federal livestock grazing allotments cover about 30 percent of the total area grazed in the U.S. (not including Alaska); however, Federal lands produced 13 percent of the total AUMs nationally. According to 1988 estimates, less than 5 percent of the nations beef cattle and 30 percent of the sheep graze on Federal lands. In western states, one third of the beef cattle is grazed at least part of the year on Federal Lands. About 2.2 million cattle and 2.1 million sheep graze on BLM allotments each year. In many cases, large (greater than 500 head of cattle) livestock operators use the public rangelands (15 percent of the operators use 58 percent of the allotments) (GAO, June 1988a and b).

[1] Includes the states of Arizona, California, Colorado, Idaho, Kansas, Nebraska, Nevada, New Mexico, North Dakota, South Dakota, Oklahoma, Utah, Washington and Wyoming.



Area	Non-Federal Grazing Lands	Federal Grazing Lands (BLM & FS)
Western U.S.	345.5 Million	260 Million
Eastern U.S.	114.6 Million	Not Available
Texas/ Oklahoma	142.2 Million	0
Total	603 Million	461 Million

*NF - Non-Federal lands, acreage based on 1982 figures.

Figure 1. Federal and Non-Federal grazing land in the United States, by Farm Production Regions.
Source: U.S. Grazing Lands: 1950-1982, Department of Agriculture.

Grazing Fundamentals

Livestock grazing on Federal lands usually involves either cattle or sheep operations. Typically, cattle are grazed in one of two types of operations, "cow/calf" or "steer." In cow/calf operations, cows and their calves are grazed until the calves are weaned to produce a calf crop. Each year, the calf crop is sold between the ages of 6 and 12 months, to feed lot operations or to other ranchers as breeding stock. A limited number of calves may be retained by the rancher to become breeding stock. Unlike cow/calf operations, steer operations are seasonal and use forage for 3 to 9 months to fatten cattle that are then sold to feedlots. Unlike cow/calf and steer operations, sheep are typically herded through allotments and graze on a seasonal basis to take advantage of more succulent and palatable forage. As the prime forage is consumed, the sheep are moved to new areas. Different species of livestock graze in different ways. Herded sheep usually use slopes and upland areas, while unherded cattle prefer lesser slopes or bottom lands. Of the forage consumed by livestock, cattle consume the most, estimated by the Bureau of Land Management and Forest Service as 87 to 89 percent of allotted Federal land forage (GAO, June 1988b). Wildlife grazing, in addition to livestock grazing, will also impact forage allotments.

When and where to graze livestock in order to optimize profits and provide ecologically-desirable results depends on many factors. Availability of forage such as grasses, forbs, or even brush is one of the prime considerations, as is easy access to water. Grazing animals prefer leaf tissue over stem tissue, and green plant material over dry material (Wallace, 1984). As would be suggested by these general rules, in some areas, streamside grazing by cattle often is more than twice the overall pasture use, with reports of riparian areas comprising less than 2 percent of the total allotments providing over 80 percent of the forage (Platts, 1986). Allotment management plans, however, can moderate this phenomenon.

Although prediction of forage growth and proper grazing may be scientifically modelled, sustainability of forage production from one year to the next depends on how heavily the area is grazed, as well as other site specific factors and variables such as annual precipitation. Most plants can withstand some loss of foliage and maintain their competitive position in the ecosystem and, in some instances, moderate grazing may increase the production of plant material. However, the approach to estimating the proper grazing intensity is complex, weighing site specific factors such as plant physiology, soils, micrometeorology, plant demography, and competitive ecology.

In monitoring grazing areas, plant vigor and species composition and diversity are major elements in determining if the area is too heavily grazed. Plant vigor reflects the capacity to rapidly produce both vegetative and reproductive shoots, the storage of nutrient reserves and effective root system volume, especially depth, when soil moisture and temperature are conducive to growth. Specific measures of vigor include numbers of tillers produced following defoliation, total plant height, leaf length, seed production, soluble carbohydrate concentrations, and root growth (Caldwell, 1984). In some cases, empirical measures are used to evaluate plant vigor. These include the ability to overwinter, to endure subsequent drought following defoliation, or to produce seed in a year following defoliation. However, less than positive results of empirical evaluations may not be known until the impact has occurred.

In general, livestock grazing can be characterized in terms of intensity, duration and timing. In a simplistic manner, grazing intensity is indicative of the amount of forage in a pasture that is grazed. Grazing intensity is measured by number of animals per unit month and ranges from light to heavy; light grazing is considered as use of 20 to 40 percent of the available forage, and moderate grazing is estimated as use of between 40 and 60 percent of available forage. The term moderate grazing also indicates that stocking rates are between those in a lightly grazed pasture and those in a heavily grazed pasture. Heavy grazing, 60 to 80 percent of available forage, is still practiced, and is considered a likely cause of poor conditions of riparian and other areas. Heavy grazing may also be defined as the amount of forage consumed in a pasture in excess of its sustainable capability. In assessing the impacts, however, much more is required than just the level of forage use. No grazing strategy is implemented the same on every allotment. Rangeland management requires the integration of complex site-specific factors, only a few of which are described here.

The timing for a first release of livestock into an area is an important factor in grazing management, sustaining plant growth from season to season, and in trapping of sediment to rebuild riparian areas. Early grazing begins when the cool season plant growth has peaked and warm season plants are beginning their growth. Early grazing ends with the flowering of key species. Late grazing is conducted only after seed ripe time when the period of maximum warm season plant growth is over and seeds have been produced; the seeds then may be trampled into the ground by livestock. Some growth of cool season plants may occur if moisture and soil temperatures allow. In order to maintain seasonal grazing, livestock are often rotated from pasture to pasture, utilizing different pastures at different stages of the growing season. Though rotation of livestock has typically been associated with heavy stocking for short durations, it has also been used for short or long periods and with light stocking.

Using these concepts, grazing systems have been developed to manage livestock. Grazing systems are plans that differ with respect to periods of grazing, intensity of grazing, season, and stage of growth of vegetation. Grazing systems are useful in that they may increase productivity of the land and, ultimately, of livestock, by controlling grazing by both wildlife and livestock. Certain specific systems have proven to be especially effective in riparian areas that are more susceptible to degradation from overgrazing. Examples of various grazing systems are provided below for descriptive purposes. Actual design and implementation of a grazing system requires the collection of site-specific data and the analysis and integration of complex site-specific variables by personnel trained in the field.

In addition, no grazing system is implemented the same on every allotment. Allotments are unique, and management can only be designed through a comprehensive, integrated approach. Management strategies are only as good as the permittee responsible for implementing the system. The best possible system will fail without the commitment from the permittee to make it work. It should not be assumed that a system will work in every situation. For example, while rotational grazing using sheep is generally a good system for riparian protection, the system may not work if the herder concentrates the sheep in streamside areas. Examples of grazing strategies are described below (Platts, 1986, 1990, and 1991).

<u>Continuous Season-Long</u>. Under this grazing scenario, livestock have unrestricted access to a specified range area for an entire vegetation growing season. Advantages are that season-long continuous grazing permits maximum forage selectivity, while minimizing disturbances to livestock by gathering, moving, and change in quality of vegetation (Platts, 1990). Drawbacks may be that livestock overgraze certain vegetation or areas before others. In addition, livestock will generally obtain much of their diet along riparian areas, typically minor portions of grazing allotments (Platts, 1986).

A 1977 study by Marcuson found that average channel width in a riparian area to be much wider after season long grazing at 0.11 ha/AUM than in a comparable ungrazed area. This study also found that heavy grazing and trampling by cattle left only 224 meters of undercut bank per kilometer in the grazed area versus 686 meters of undercut bank per kilometer in the ungrazed area. As a result of these erosional impacts to riparian areas under this grazing scenario, Platts does not consider this strategy to be useful in those areas, as fishery productivity would be seriously impacted.

Short Duration - High Intensity. Short duration, high intensity grazing generally describes high stocking, high intensity use in a designated area, over a short period of time. Livestock are placed in an area for a period of one day to several weeks before being moved to the next area. This type of strategy requires numerous pastures in order to ensure that a grazed section is unused for a significant amount of time to permit regrowth. The layout of pastures is sometimes subdivided to resemble a "wagon-wheel." This method requires almost daily checks on vegetative conditions to prevent overuse. In general, this method is out-dated and is infrequently used.

Three Herd - Four Pasture. Also referred to as the Merrill Pasture System, this strategy allows each pasture a period of nonuse within one four year cycle. Useful in upland areas, the Merrill Pasture System requires less animal movement than other heavy use strategies, and has succeeded in generating higher plant productivity in conditions with sufficient precipitation. However, one four-month period of nonuse over a four year period is not sufficient to rehabilitate a heavily impacted riparian area.

Seasonal Suitability. This strategy requires substantial fencing and frequent movement of animals from pasture to pasture, providing heavily used areas with periods of nonuse for regeneration, during selected periods of the grazing season. Depending on the extent of use prior to periods of nonuse, riparian areas may not be able to regenerate sufficiently before livestock are re-introduced to the area. In addition, there is seasonal variation in streambank stability, with greater potential for erosion during the dryer hot season.

Holistic Method. This grazing strategy may be less straight-forward than others, requiring training and management skills to enable heavy stocking and frequent movement dependant upon the growth cycle of plants and other environmental factors. This method also utilizes livestock as a soil churning mechanism to break up the soils, and increase soil porosity (its effectiveness is under debate). While upland areas may benefit from this type of management, this grazing method may erode streambanks in riparian areas, impacting streamside vegetation and overall riparian habitats.

Deferred. Deferred grazing strategy defers grazing in one or more pastures to permit desired growth or regrowth or to produce ripe seeds prior to being grazed. The period of deferment may continue for several years to allow vegetation to reestablish itself. This grazing strategy requires a substantial amount of fencing and cattle movement, though the periods of rest offer opportunity for regrowth of preferred grazing vegetation. Deferred rotation in a riparian area may be a useful grazing strategy in a riparian area if overstocking is prevented in order to avoid streambank shear and erosion.

Deferred Rotation. The deferred rotation strategy delays grazing of key species until seeds have matured by systematically rotating livestock among a number of pastures. If one pasture is grazed early one year, pasture use sequence would change the following year so that a different pasture was grazed early. This method requires a fair amount of fencing, however, vegetation is able to store carbohydrates and set seed every other year. The period of nonuse will vary throughout the each year, allowing areas of nonuse during critical periods to allow plant cover to increase.

Stuttered Deferred Rotation. Similar to the deferred rotation strategy, one pasture is deferred for part of the plant growth period. The deferment is passed on to a different pasture but in the stuttered method grazing use occurs on one pasture early for the first two years and another late the following two years, whereas deferred rotation changes every year. A great deal of fencing, and movement of livestock is required under this grazing scenario. However, as with the use of Deferred Rotation, brushy species are given an opportunity for regrowth.

Rest-Rotation. This grazing strategy involves rotating livestock from one range area to another in order to prevent overgrazing. Though this method may be costly since it may require fencing to carve out range areas within an allotment, it allows grazed rangeland to rehabilitate while cattle are occupying another portion of an allotment. This strategy has shown measurable success in some habitats.

The rest rotation strategy is a multi-pasture design strategy that provides at least one year of rest for a grazed pasture. This strategy is frequently combined with deferred, early, and late grazing techniques so that pastures are rested until seed ripe time, and rested for seedling establishment. Depending upon vegetation types and soil moisture content and temperature, three or more pastures are needed for rest rotation to be successful.

Double Rest-Rotation. Under this strategy, an area or pasture with the highest riparian and stream values would receive twice the amount of rest compared to the amount of rest allocated under the normal rest-rotation grazing cycle. In a three pasture system, the most valuable riparian-stream area would receive 2 years rest. A Forest Service study of a double-rest-rotation system, graze early then rest 2 years, then graze late and rest 2 years, showed no adverse riparian-stream impacts.

Rest-Rotation with Seasonal Preference. This strategy is most often applied to sheep since this method requires frequent movement of the livestock in response to signs of range, riverine or riparian habitat deterioration. The strategy encourages use of areas during periods of least impact to vegetation, allowing plants to be grazed at particular times to allow rest to recover from past grazing use.

Riparian Pasture. This grazing strategy places the riverine-riparian system within a controlled unit, to permit grazing only in those areas of the stream that can provide vegetation without being negatively impacted. Additional fencing is required under this scenario to prepare riparian pastures that encourage utilization of both riparian and upland areas. Overuse of upland areas of the pastures is also a concern in the event of increased sediment, or overland flows impacting the stream. The advantage of individual pastures is the ability to encourage distribution evenly within each pasture.

Seasonal Riparian Preference. As with the Riparian pasture method, use of this strategy encourages grazing of plants and streambanks during periods when the vegetation is less vulnerable to sustaining damaging impacts. Fencing and frequent animal movement are also necessary in order for this strategy to be successful, and grazing within each pasture must happen over a narrow period of time.

Winter. A form of seasonal grazing, winter grazing takes place when range vegetation is dormant and streambanks frozen. Impacts to riparian areas may diminish under these conditions, since streambanks tend to be more capable of withstanding the impacts of hooves while frozen. In riparian areas, winter grazing in areas of low temperatures but little snow can be beneficial to the extent that streambanks are sturdier, and vegetation dormant.

Holding. The holding strategy is a short to long term method of containing livestock in a specific area of land prior to moving them. This strategy permits animals freedom to move within a designated area. These holding areas are useful not only to allow other pastures to be prepared for grazing, but can also be used as disease treatment facilities, and for breeding purposes. Pros and cons associated with this grazing strategy are similar to those under the season long continuous strategy, such as preferred plants and riparian areas receiving excessive use (Platts, 1990).

Corridor fencing. Stream corridor fencing in riparian areas prevents overuse of streamside vegetation, and assists in the rehabilitation of denuded portions of a riparian zone. This strategy usually requires extensive fencing and involves high maintenance costs.

Rest. Certain areas may be rested until vegetation and/or riparian habitats are permitted to re-establish themselves and regrow.

Rangeland Management

Modifications to rangelands can be used to mitigate impacts of livestock and wildlife grazing and are discussed in a later section on mitigation. While modifications to rangeland can enhance grazing opportunities, modifications may also result in adverse effects on water quality, as well as aquatic and terrestrial ecosystems, if not properly planned and managed. Platts (1991) alluded to the variety of activities that could occur as part of rangeland management, including the fertilization of lands; irrigation and drainage of wetlands; brush, forb, and pest control; debris disposal; mechanical treatment of the soil; seeding, prescribed burning; water supply development; fencing; and timber thinning. Depending on the frequency, extent and appropriate implementation of these range improvement practices, both positive and negative effects can occur. Potential negative impacts include erosion and sedimentation, hydrologic modification, chemical contamination (pesticide and fertilizer), and unfavorable ecosystem alteration. However, if rangeland improvements are tied to the attainment of specific resource objectives, then such improvements may reduce the severity of grazing impacts, thus the implementation of sound grazing practices.

POTENTIAL SIGNIFICANT ENVIRONMENTAL IMPACTS

Both livestock and wildlife overgrazing may cause direct impacts resulting in physical changes to the rangeland, such as the removal of protective plant cover and damage from hoof action and trampling to ground surfaces. These direct impacts may contribute to a host of indirect impacts such as erosion and stream channel modification. Both direct and indirect physical impacts often result in changes to terrestrial and aquatic ecosystems. These changes to the rangeland from overgrazing occur in both upland and riparian areas. Impacts in both environs can affect stream water quality, although activities in the riparian zone often cause more immediate and severe impacts. While it is difficult to make generalizations concerning the effects that livestock and wildlife grazing practices have on rangeland due to the geographic variability of vegetation, soils, climate, and topography, the majority of the research reviewed for this document points out some common trends. To fully assess the applicability of these trends, a knowledge of the site-specific conditions is important. Even the grazing species is important; cattle and sheep have different impacts on streambanks. The stream and its watershed function as a unit and therefore, management is most effective on a basin-wide approach (Platts, 1986). Because much Federal land is intermingled with private land in a checkerboard pattern, it is important to plan for the total ecosystem, considering grazing activities on adjacent and nearby private land, as well as the activities on Federal land. For example, overgrazing on private land upstream of public land may cause impacts to the public land. Although the land manager's administrative responsibility does not apply on private land, recognizing impacts on a watershed basis and integrating these into grazing management strategies is important.

One of the more significant hydrologic and water quality effects associated with overgrazing results from impacts on soil from livestock hoof action and trampling. For example, hoof action and trampling can disrupt natural soil conditions (e.g., soil structure, bulk density, and permeability) and cause soil compaction, which leads to increased runoff and associated soil erosion and loss. The removal of plant cover by the grazing animals exacerbates these problems by leaving even more soil bared to disruption and compaction. Also, the removal of plant cover by grazing animals frequently changes the overall density and composition of the native vegetation. As grazing-related activities create conditions that increase runoff and soil erosion from the rangeland, stream water quality is primarily affected by the increased amount of sedimentation. Also, hydrologic changes to the stream channel due to increased water velocity and flow can occur. The reduction in plant cover can indirectly affect water temperatures, especially expanding the range of temperatures experienced in the stream and increasing maximum temperatures. Compaction can also affect the ability of vegetation to establish, thus exacerbating erosion.

The effects caused by overgrazing result from a variety of interrelated factors such as climate, vegetation, topography, soil characteristics, and the intensity, type and duration of livestock and wildlife grazing. Therefore, the nature and extent of impacts from overgrazing will vary from location to location due to the normal variability of ecosystem specific factors. Despite these variabilities, the mechanisms causing the impacts (e.g., soil compaction and increased runoff) are similar. Impacts can also vary significantly between grazing strategies. Because activities throughout a stream's watershed (i.e., upland and riparian areas) can affect stream water quality, grazing strategies should address both areas.

Livestock and wildlife grazing activities are associated with other causes of surface water degradation such as bacterial/fecal contamination of water bodies, stream bank erosion and modification associated with hoof or head (scratching, butting or digging) action, withdrawal of water for irrigation of grazing areas, and drainage of wet meadows.

Figure 2 illustrates some of the interrelated impacts that stem from livestock and wildlife foraging and trampling, such as changes in vegetative cover (density and type), affecting physical soil condition or surface water hydrology. In general, the adverse effects associated with grazing increase as the intensity of grazing increases.

This chapter is divided into two major sections: Direct Impacts and Indirect Impacts. Indirect Impacts are further divided into physical impacts and ecosystem impacts. The major direct effects includes a description of the effects of overgrazing and livestock trampling on vegetation and ground surface conditions and the ensuing changes to physical characteristics of the rangeland, and changes to infiltration rates. The discussion of the indirect impacts addresses erosion and sedimentation, channel modification, water table changes, bacterial contamination, and temperature changes. While not all grazing results in adverse impacts, and there may be some favorable impacts that are the result of grazing, this section focuses on the potential adverse impacts of grazing activities.

Direct Impacts

Overgrazing of livestock and wildlife can affect rangeland in two major ways: (1) by reducing the density (i.e., percent-cover) and quality of vegetation, and (2) by disrupting soil conditions and causing soil compaction by hoof action and trampling. Each of these effects creates conditions which lead to increased surface water runoff, sedimentation, and erosion. Livestock foraging reduces the amount of cover provided by vegetation (including plant litter), which in turn creates a situation where soil compaction, reduced rainfall infiltration, increased runoff, and soil erosion can occur. The trampling by livestock further compacts soil, reducing infiltration and increasing surface runoff and resulting soil erosion. (Blackburn, 1984 and Kauffman and Krueger, 1984)

Vegetation. Livestock overgrazing can reduce the health and vitality of rangeland vegetation, therefore, reducing the amount of ground cover provided by the vegetation. Vegetation is specifically affected by livestock in the following ways:

- trampling causes soil compaction, thus decreasing water infiltration, causing increased runoff, and decreased water availability to plants;
- herbage is removed, which allows soil temperatures to rise and increases evaporation to the soil surface;
- physical damage to the vegetation occurs by rubbing, trampling, and browsing (Kauffman and Krueger, 1984).

An additional factor is that as foliage is removed, plants put a greater portion of energy into regrowth of leaves and less toward root growth which has the effect of reducing root biomass which in turn reduces soil stability and leads to increased erosion. Altering vegetation patterns can result in greater susceptibility to draught, fire, insects, and exotic plant competition.

As vegetation is harvested, total plant density and cover may decline, and a compositional change may occur (e.g., decrease of grasses and forbs and increase of sagebrush). In some cases, less desirable species may result. By altering the amount of vegetative cover and composition, overgrazing ultimately increases the amount of bare soil on the rangeland that is subject to runoff and erosion. It also creates conditions that can modify stream temperatures, thus causing a host of ecological changes. Also, changes to vegetation from overgrazing can often result in an overall decrease in the grazing capacity of the rangeland.

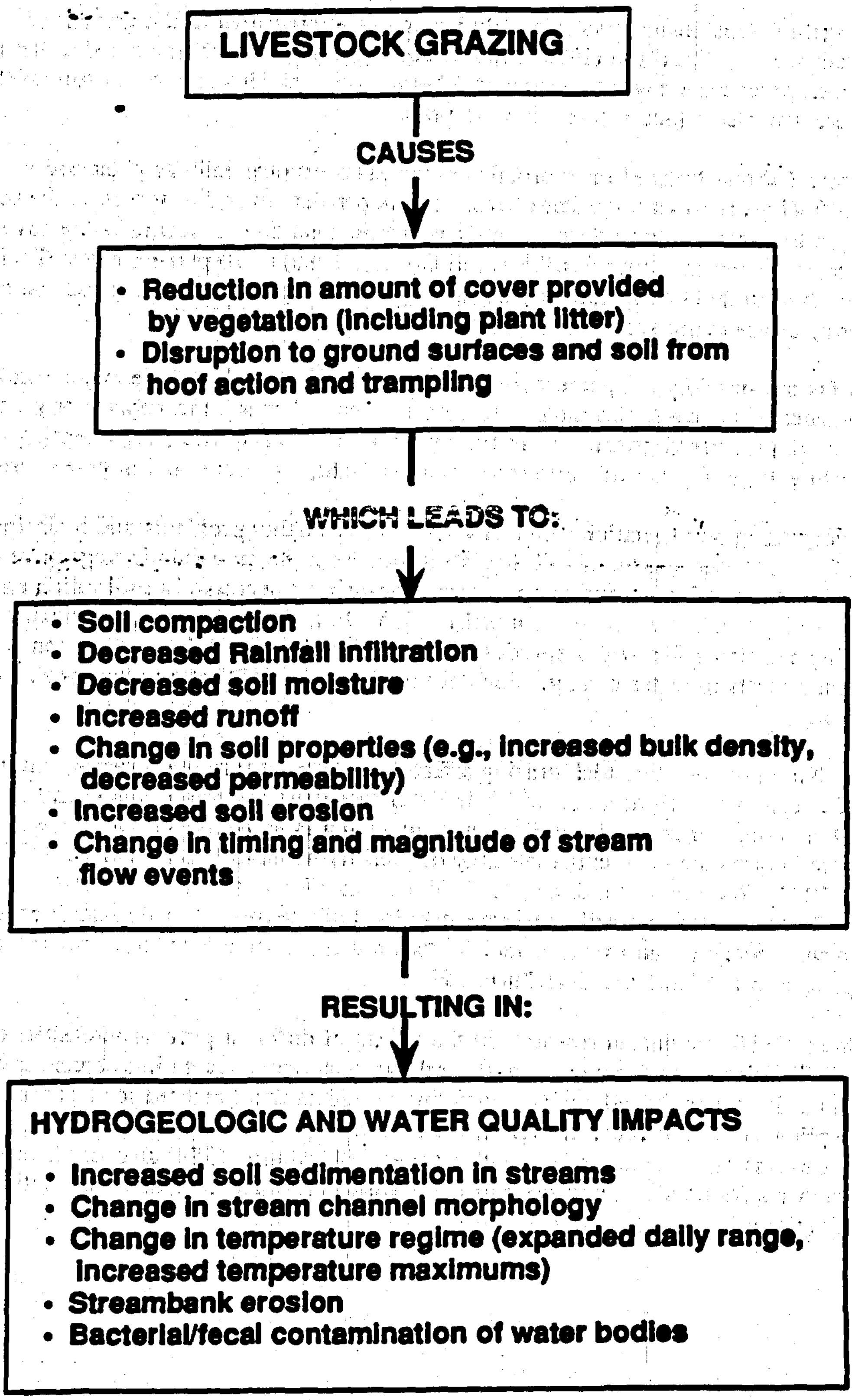


Figure 2. The Interrelationship of Grazing Impacts.

Impacts to the rangeland (and ensuing water quality impacts) are intensified as the amount of vegetative cover decreases. Blackburn (1984) summarized two studies which attempted to define a cover threshold (i.e., percentage cover by vegetation) below which serious impacts to soil infiltration and associated increased runoff (and soil erosion) occurred.

For example, Figure 3 shows that sediment production increases exponentially as plant cover decreased. These findings represent one study area, and the percent cover that serves as the threshold point varies with location according to a variety of site specific conditions. Generally the cover thresholds range from 50 percent cover (Dadkhah and Gifford, 1980) to 70 percent cover (Packer, 1953). However, the threshold point can vary depending on the initial amount of vegetation at the site and the intensity of use at the site.

Grazing intensity (as measured by the percentage of ground trampled) is one of the major factors that affects the maintenance of the cover threshold. As common sense dictates, the impacts of grazing on vegetation increase with increased grazing intensity; high intensity grazing (i.e., high density) causes serious impacts, while there may be little difference between light, moderate, and ungrazed areas.

The impacts of overgrazing on vegetation result in surface water quality problems and hydrologic modification largely due to the amount of soil that is exposed from the reduction in vegetative cover. This can increase the impact of raindrops on soil, possibly causing a decrease in infiltration rates, increase in surface runoff, and/or an increase in soil erosion. In a similar manner, livestock hoof action and trampling can also affect soil properties and ground surface conditions which can cause a range of subsequent impacts to water quality. Each of these impacts (infiltration rates, sedimentation) are described below.

<u>Infiltration Rates</u>. Not only does livestock grazing affect the rangeland through foraging, but the hoof action and trampling causes soil compaction which leads to decreased infiltration rates, and increased runoff, and/or soil erosion. Innumerable studies have shown that infiltration rates decrease as a result of trampling. These impacts increase as the intensity of grazing increases (Warren et al., 1986; Wood and Wood, 1988; Wood and Blackburn, 1981; Weltz and Wood, 1986). The most important factors affecting infiltration rates are: soil aggregate stability, bulk density, organic matter content, and initial soil moisture content; and extent of mulch, standing crop, ground cover, perennial grass cover, and total grass cover (Wood and Blackburn, 1981).

Dadkhah and Gifford (1980) conducted research on the effects of different grazing intensities on infiltration rates. Infiltration rates decreased significantly with increased trampling percentages up to 40 percent trampling. In this study, 40 percent trampling served as the threshold for infiltration reductions; at trampling rates 40 percent or higher, the researchers found no significant differences in infiltration rates regardless of the extent of vegetative cover. Blackburn (1984) also summarized a number of infiltration studies conducted on the Northern Great Plains that compared infiltration rates to grazing intensity (Table 1).

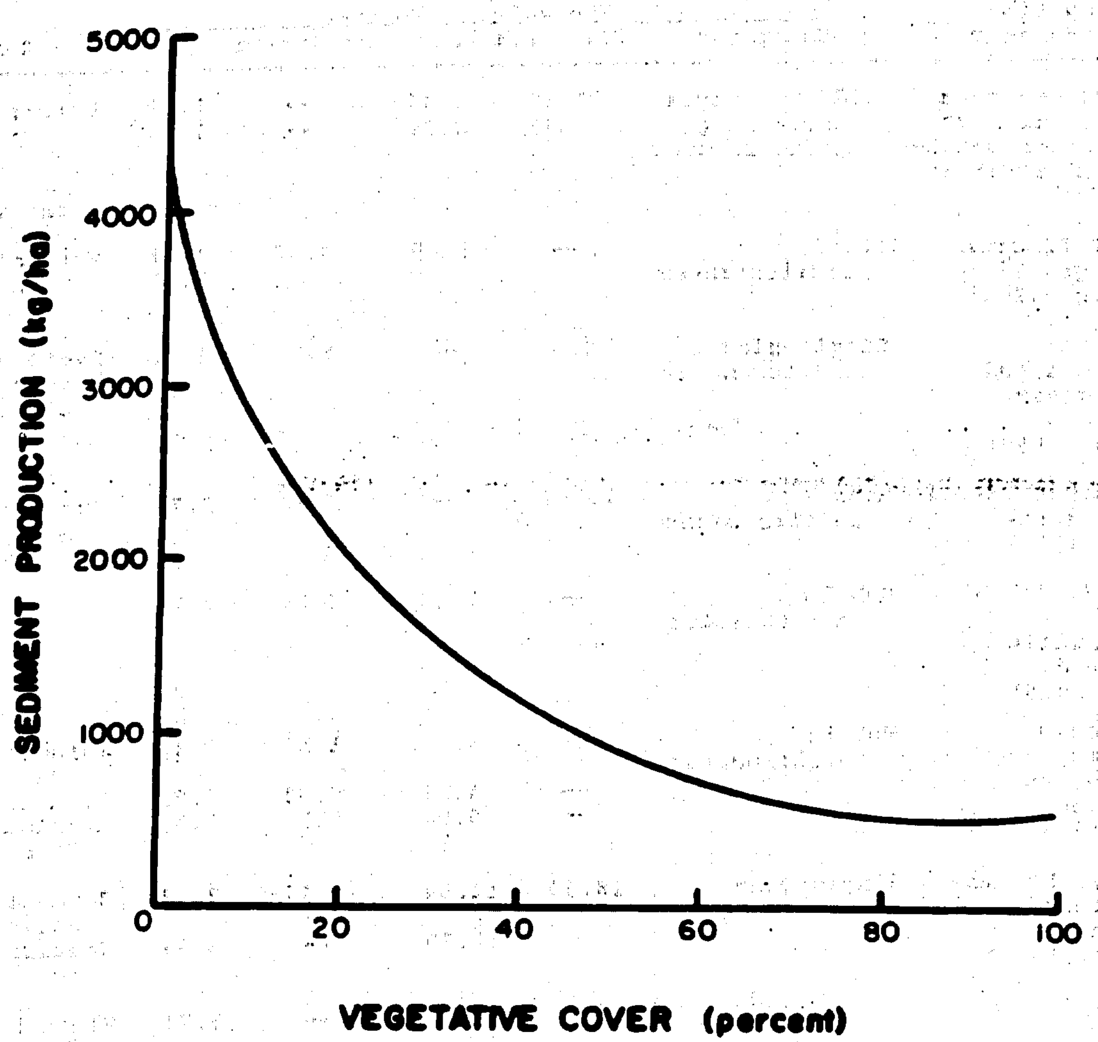


Figure 3. Sediment production as a function of vegetation cover*.
Source: Dadkhah and Gifford, 1980.
*** will vary widely depending on geography, soils, climate**

Table 1. Summary of studies of the influence of livestock grazing on infiltration on the Northern Great Plains.

Study Site and Reference	Equipment	Infiltration Capacity (cm/h) by Grazing Intensity: Ungrazed	Light	Moderate	Heavy	Remarks
Fort Peck, Montana Nuttall saltbush and crested wheat-grass (Branson et al., 1962)	USGS tube-type sprinkling infiltrometer	0.65	0.45	--	0.92	Unfurrowed
		3.02	2.29	--	1.10	Furrowed, seeded averaged over soil type and years
Southwest Alberta Fescue grassland (Johnson, 1962)	Mobile infiltrometer	--	5.69	4.06	4.14 3.53	Very heavy grazing
Hays, Kansas Blue grama and Buffalograss (Knoll and Hopkins, 1959)	Single-ring infiltrometer	6.55	--	5.28	4.01	Exclosure had not been grazed for 13 years
Mandan, North Dakota Mixed Prairie (Rauzi, 1963)	Mobile infiltrometer	10.84	--	6.10	3.76	Exclosure had not been grazed for 21 years
Cottonwood, South Dakota Mixed Prairie (Rauzi and Hanson, 1966)	Mobile infiltrometer	--	7.49	4.24	2.76	
Nunn, Colorado Blue grama and Buffalograss (Rauzi and Smith, 1973)	Mobile infiltrometer	--	1.40	1.14	1.27	Shingle sandy loam
		--	4.32	5.33	2.03	Nunn loam
		--	5.00	5.13	2.03	Ascalon sandy loam
Miles City, Montana Mixed Prairie (Reed and Peterson, 1961)	Single-ring infiltrometer	18.58	11.04	10.96	7.19	Blue grama upland
		--	12.29	--	5.69	Western wheat-grass bench
			17.12	--	6.74	Western wheat-grass bench
Western North Dakota Mixed Prairie (Whitman et al., 1964)	Single-ring infiltrometer	15.24	--	--	7.87	

While there was some variability among the results due to site-specific conditions and variations in study methodology, the following general trends were noted for all of the research evaluated:

- Differences between light and moderate grazing were usually very small.
- Heavy grazing almost always caused a reduction in infiltration rate.
- Soil bulk densities appeared to increase with grazing intensity and were higher on grazed pastures than on ungrazed pastures.

Some researchers have attempted to examine infiltration rates in the context of different grazing strategies. In general, these findings supported the above assertions that as stocking intensity and density increase, infiltration rates tend to decrease. Wood and Blackburn (1981) noted that infiltration rates in deferred-rotation treatments approached the near-optimum infiltration rates demonstrated in the grazing exclosures and exceeded those in the heavily stocked, continuously grazed treatment. Infiltration rates in a high intensity, low frequency (HILF) treatment were similar to those of the heavily stocked, continuously grazed treatment (Figure 4). Research by McGinty, et al. (1978) also found that infiltration rates for a pasture subject to a 4-pasture deferred-rotation grazing system were similar to those of a 27-year exclosure, while infiltration rates were significantly lower for a heavily, continuously grazed pasture.

Indirect Physical Impacts

The previous section described how poor management of livestock grazing may create conditions that can decrease infiltration, increase runoff, and increase sedimentation and erosion from rangelands. These direct impacts can affect the hydrologic regime and water quality of receiving streams, ranging from channel modification to problems associated with sedimentation. The following section describes some of these indirect impacts, including sedimentation, channel modification, changes in the water table, bacterial contamination, and changes to a stream's temperature regime.

Erosion and Sedimentation. The decrease in infiltration normally associated with increased grazing intensities results in an increase in overland flow. This increase in runoff (especially volume and velocity) often results in increased erosion and sediment production. Also, the loss of vegetation resulting from livestock grazing leaves more ground bare further exacerbating the sedimentation problems associated with grazing. As mentioned earlier, Dadkhah and Gifford (1980) found that sediment yield increased exponentially as the amount of plant cover decreased.

Lusby (1979) conducted extensive research on the effects of overgrazing on the hydrology of salt-desert shrub rangeland in west central Colorado. Runoff and sediment were measured in reservoirs at the lower end of grazed and ungrazed reservoirs and watersheds. Runoff from grazed watersheds averaged from 131 to 140 percent of that from ungrazed watersheds from 1954 through 1966. Sediment yields during the same time period ranged from 134 to 196 percent of that from ungrazed watersheds.

Studies examining sediment production as function of grazing intensity generally echoed the results of the studies examining infiltration rates, finding that sedimentation increases as grazing intensity increases. Wood and Blackburn (1981 a,b) conducted research examining the effects of various grazing strategies on sediment production, as well as a number of other physical parameters at the Texas Experimental Ranch. Table 2 summarizes these results. Wood and Blackburn (1981a) found that sedimentation rates from the heavily stocked, continuously-grazed pastures and the HILF pasture exceeded those of the deferred-rotation pastures and exclosures at the site in Texas.

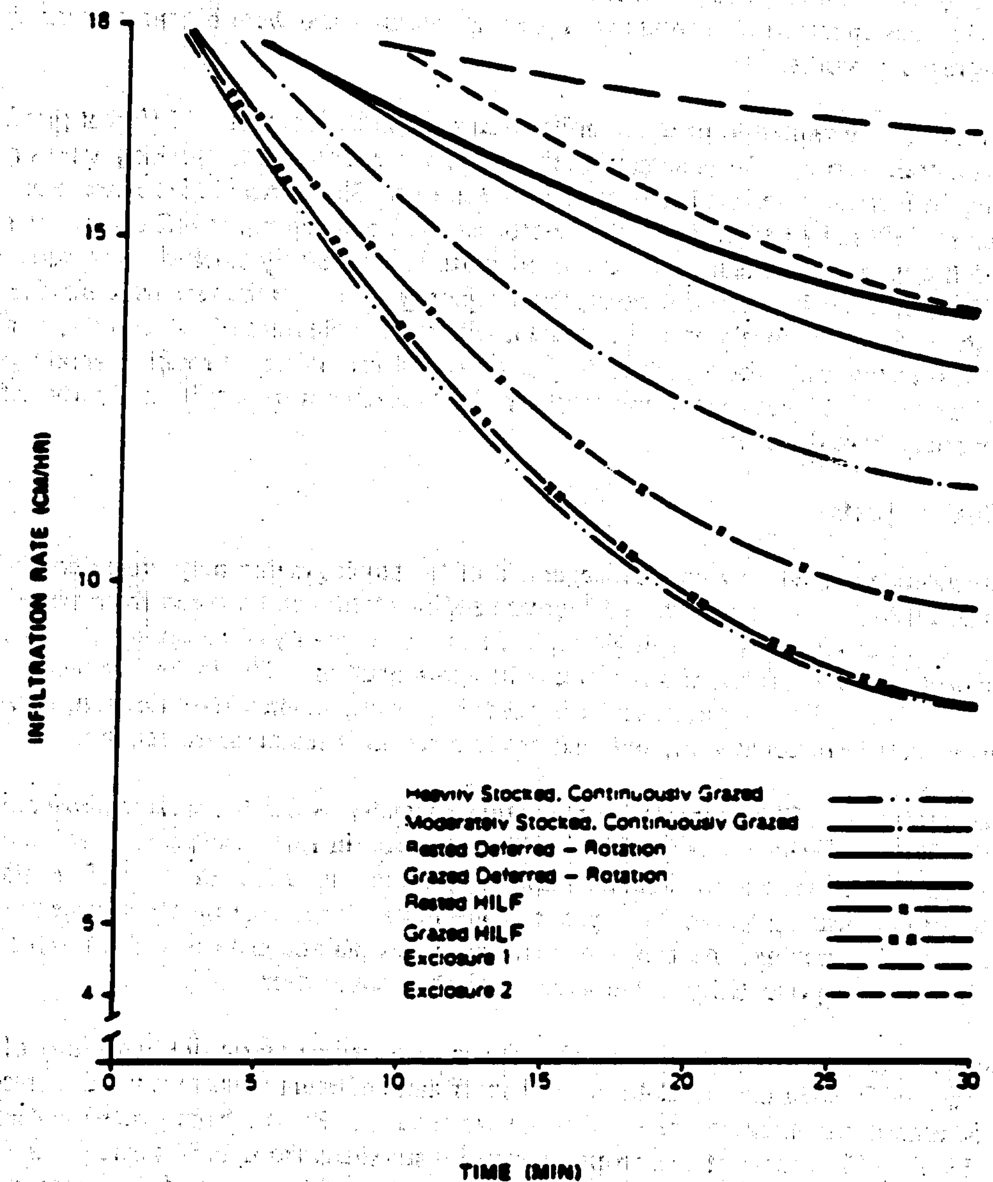


Figure 4. Mean infiltration rates of the midgrass community for various grazing practices at the Texas Experimental Ranch. Source: Wood and Blackburn, 1981a.

Grazing Treatment	Grass Standing Crop (kg/ha)	Mulch (ton/ha)	Bare Ground (%)	Bulk Density (g/cc)	Organic Matter (%)	Aggregate Stability (%)	Infiltration rate after 30 min (cm/h)	Sediment Production (kg/ha)
Heavy continuous	1508 d [1/]	1.2 d	25 a	1.8 a	2.6 c	35 d	8.1 c	115 a
Moderate continuous	3333 abc	4.5 bc	6 b	1.6 b	3.7 b	48 bc	11.4 bc	28 abc
Rested deferred-rotation	3865 ab	5.1 bc	1 b	1.6 b	5.5 a	57 ab	13.1 ab	10 c
Grazed deferred-rotation	2894 c	6.1 b	5 b	1.8 a	4.1 b	56 ab	13.9 ab	14 bc
Rested HILF	2437 c	3.2 cd	17 b	1.9 a	4.3 b	60 a	9.6 bc	28 abc
Grazed HILF	2414 c	4.5 bc	17 a	1.9 a	3.5 b	45 c	8.2 c	39 ab
Exclosure 1	4569 a	12.2 a	1 b	1.3 c	4.3 b	62 a	16.5 a	4 c
Exclosure 2	4243 a	11.5 a	4 b	1.8 a	2.3 c	39 cd	13.9 ab	17 bc
All treatments	2988	6.1	9	1.7	3.8	50	11.6	32

[1/] Means followed by the same letter within each column are not significantly different at the .05 level of probability.

Table 2. Watershed parameter means for the midgrass interspace areas in each grazing treatment at the Texas Experimental Ranch.
Source : Wood and Blackburn, 1981a, 1981b.

Weltz and Wood (1986) also conducted research supporting the above assertions. At a study site in central New Mexico, they asserted that total sediment production was greater on all grazed treatments than on the exclosure. Doubling the stocking rate and applying a short-duration system resulted in significantly greater sediment concentrations and total sediment production. The researchers attributed these findings to the changes in vegetation to a less desirable weedy condition, a decrease in the amount of litter load, and an increase in bare ground resulting from overgrazing. Overall, the researchers concluded that after rangelands were grazed in a short-duration paddock the soil surface was susceptible to accelerated erosion, whereas scattering the cattle over a larger area created problems with distribution and herd control, but seemed to have lower risks of environmental damage as expressed by soil erosion, at least in the short-term.

One of the primary impacts of livestock overgrazing to surface water bodies is the increase in sedimentation associated with grazing activities (e.g., vegetation removal, trampling). The increase in runoff and sedimentation from rangelands can significantly increase sediment loads in water bodies. This can result in many serious water quality impacts, particularly those relating to the health of the aquatic ecosystem. The water quality impacts associated with sedimentation are discussed in more detail in a later section of this document on aquatic ecosystems.

Channel Modification. As described in the previous section, the impacts of livestock overgrazing associated with vegetative removal and trampling can create conditions (i.e., bared and compacted soil) which may result in increased volume and velocity of runoff and increased peak flow discharges. This input of additional runoff water into streams can result in fairly significant channel modification and a host of related effects (e.g., reduction in the cover and area suitable for fish habitat). Depending on soil and subsurface conditions, these rapid adjustments may take two forms: excessive downcutting or incision, including head-cutting (not just down cutting, but cutting back upstream as well), or excessive lateral or sideward migration of the stream (Bureau of Land Management, 1990).

Incised channels typically occur when the stream is in early stages of development and/or is characterized by unresistant bottom materials. For example, channels in fine, deep alluvial soils are prone to incision. They result from either downstream base-level lowering or localized gullying initiated by increased runoff rates and/or lowered resistance to erosion. This type of deep channel incision can result in the following two important changes in the local stream environment, particularly in riparian areas: (1) advancing gully systems increase peak discharge making the stream very efficient at scouring channel beds and banks and transporting sediment, and (2) degrading channel beds produce a drop in the local water table therefore creating a water stress on the riparian vegetation. The subsequent loss of riparian vegetation further exacerbates hydrologic changes. For example, it may result in an even lowered resistance to surface runoff and higher flow velocities during flood events.

Channels will widen and become laterally unstable if stream bottoms are comprised of relatively resistant materials. For example, coarse alluvial channels or channels with structurally controlled beds tend to respond to increased runoff and flow by becoming wider and shallower with less steep banks. Channels that are laterally unstable may be less capable of carrying high flows and thus can cause serious riparian damage by bank cutting or channel realignment during times of high flow. Increased sedimentation from upstream sources can greatly exacerbate these effects (Bureau of Land Management, 1990). An illustration of the channel changes is shown in Figure 5.

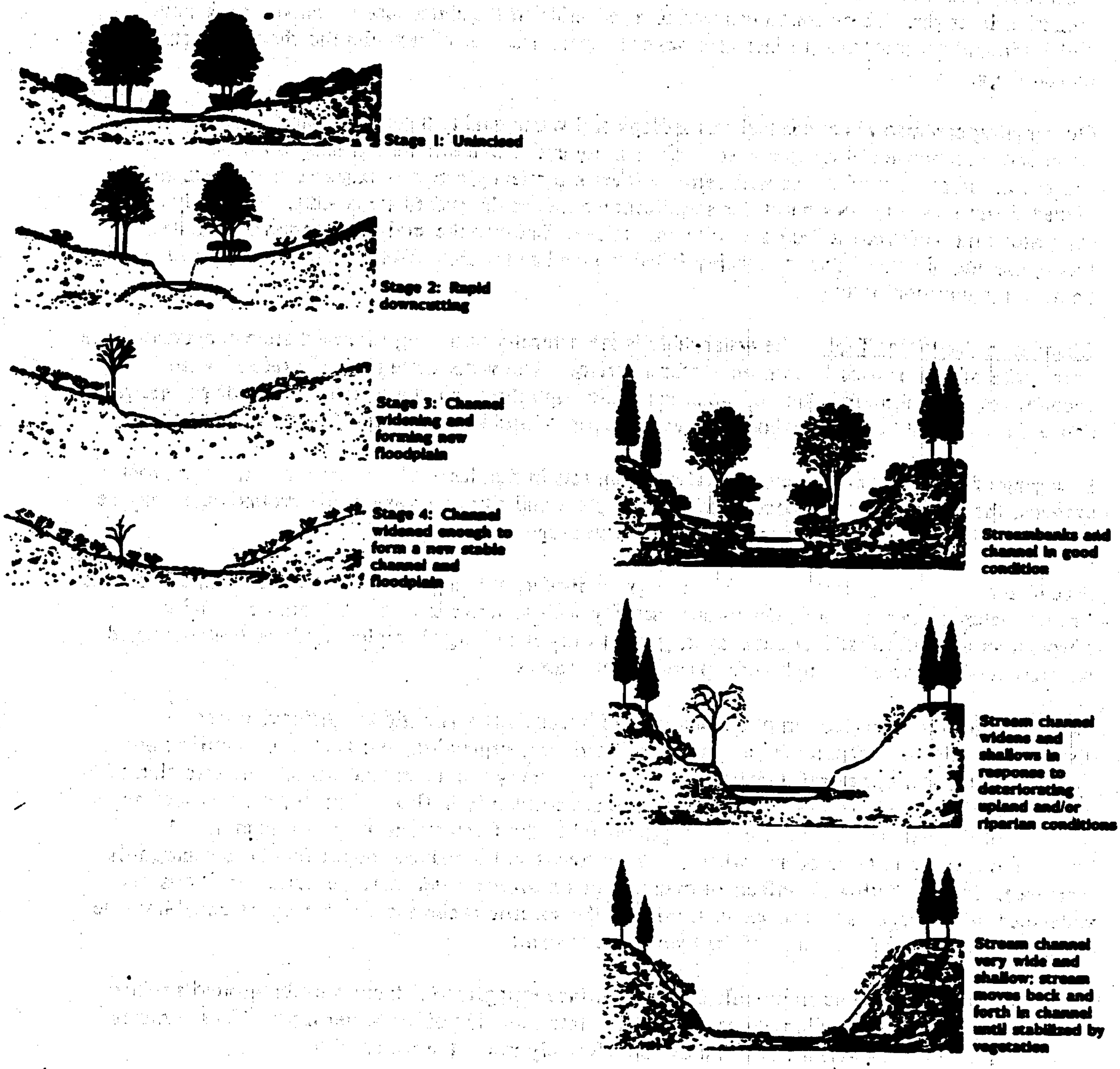


Figure 5: Stream Channel Morphology
Source: "Livestock Grazing on Western Riparian Areas"
Northwest Resource Information Center, Inc., July 1990.

Hubert et al. (1985) examined the impact of various grazing strategies and intensities on the hydrologic conditions of streams. The study examined selected stream parameters (e.g., width) and noted the range of responses to light versus heavy grazing (Table 3). The data showed that, for the most part, intensive grazing caused the widening and shallowing of streams and a subsequent reduction in cover. These conditions lead to a reduction in the abundance of native brook trout, which the authors attributed to increased water temperatures associated with the changes in stream morphology.

Overgrazing can also affect channel morphology and water quality through impacts to stream banks. Bohn and Buckhouse (1986) compared bank stability under five different grazing options. They found that the amount of streambank retreat differs statistically between ungrazed treatments and grazed treatments, but does not differ significantly between the grazed treatments. The study also suggested that bank retreat increases with animal use. Because the study was somewhat limited in scope, the authors stated that it probably failed to simulate the full effects of large-scale cattle grazing on stream bank morphology.

Changes in the Water Table. The water table is the naturally occurring saturated zone contained in the pore space of soil or rock beneath the ground surface. The water table typically refers to the first encountered or shallowest saturated water zone, although there may be isolated lenses of groundwater above the water table. Deeper bodies of water occur as aquifers or isolated lenses of groundwater.

Lowering of the water table may have adverse impacts in that less water is available for plant root systems, the local hydrologic conditions are disrupted, and any other use of the groundwater may be affected such as availability for irrigation or human usage.

Precipitation is the principal source for most groundwater, although groundwater may also come from surface water (stream or lake), agricultural activity such as irrigation, or other human activity. Through an unconfined soil or rock layer, groundwater is recharged (replenished) by the downward infiltration of rainwater through pore space in rock masses.

Factors influencing the location of the water table include site and regional geology, water distribution, climate and precipitation, soil characteristics, vegetation, and land use. Aquifers are dynamic systems with natural fluctuations occurring, usually, on a seasonal basis. The direction of groundwater flow and the depth from the surface are constantly in flux. Human activities such as pumping of groundwater wells or crop irrigation add to the fluctuations in the water table. A lowering of the water table occurs when the input (recharge) is reduced or the output (discharge) is increased. In considering the effects of overgrazing on groundwater or water table conditions, the watershed or drainage basin and its uses, not just the specific rangeland, must be considered because of the complex interrelationships of the hydrologic system.

Because water tables are strongly influenced by surface topography, changes in the ground surface affect the level, quantity, volume, occurrence and flow direction of the water table. Thus, grazing activities that affect the surface topography can adversely affect the water table.

In discussing the effects of overgrazing, there are two geographic zones to consider. First, there is the broader regional upland area, then the more localized riparian stream bed area, which is composed of the stream itself (water column), the stream channel, and the banks of the stream. Beyond and above the banks is the flood plain, which forms an intermediary area between the uplands and the stream zones.

Table 3. Mean Values of Stream Habitat Variables Measured in Heavily and Lightly Grazed Reaches of Pete Creek in 1984.

Variable	Mean Value (n = 3)	
	Heavily Grazed	Lightly Grazed
Width (m)	2.9	2.2*
Depth (m)	0.07	0.11*
Width/depth ratio	43	21
Coefficient of variation in depth	47.3	66.6*
% greater than 22 cm deep	9.0	22.3**
% silt substrate	35	52
% gravel substrate	35	31
% rubble substrate	24	14
% bedrock-boulder substrate	1	3
SRI/CSI	112	110
% overhanging bank cover	2.7	30.0*
% overhanging vegetation	0.0	11.7*
% shaded area	0.7	18.3*
% bare soil along banks	19.7	13.3
% litter along banks	7.0	6.0

* indicates statistically significant difference at p 0.05
** indicates difference at $p \leq 0.10$

In both the uplands and riparian stream zones, overgrazing can adversely impact the water table. Direct effects of upland grazing are loss of vegetation, compaction of soil, and increased runoff (with subsequent decrease in infiltration). Bare soil is exposed to greater evaporation of soil moisture. Stream impacts include all of the upland impacts, plus physical degradation of the stream banks. These effects combine to cause greater erosion of the stream channel. Increased runoff, greater sediment load, sloughing of stream banks, loss of ground cover, and loss of root biomass all contribute to the instability of the stream system causing increased incision (down cutting and head or back cutting) and widening of the stream channel. Changes in the channel morphology may impact groundwater by altering the direction and rate of groundwater flow and the depth to groundwater. Downcutting lowers the streambed and the groundwater table.

Depending on site-specific conditions, groundwater may regularly or periodically flow from the subsurface strata (water table) into stream beds, adding water to the stream flow. Such conditions would add to the vitality of the stream life. Groundwater seeps from the stream banks or up from the bottom into the stream. Conversely, water may discharge from a stream to the water table.

Lowering of the water table may significantly reduce or halt water flow into a stream thus accentuating stream degradation. Physical degradation of stream banks by livestock can alter the flow of groundwater and reduce discharge to streams by compacting the soil or otherwise altering the water flow.

Another adverse impact of lowering the water table is the potential effects on plants. Roots obtain their necessary moisture through capillary action that draws water (moisture) upwards through the soil to the root zone where it is available for plant use. Excessive or improper grazing activities may cause greater evaporation of soil moisture by denuding the ground of vegetative cover and increasing soil temperature, thus drying out the soil and leaving insufficient moisture needed for plant life.

Bacterial Contamination. Livestock grazing can also cause increases in the level of bacterial pollutants (i.e., fecal coliform) in water, as well as nutrient enrichment. The level of severity is related to the intensity of grazing activities and the proximity of animals to the water. Tiedemann et al. (1988) presented research results suggesting that increasing the intensity of cattle grazing can increase the amount of fecal coliform (FC) in water to very high and potentially problematic levels. In their research, Tiedemann et al. (1988) measured concentrations of fecal coliform weekly during summer 1984 in streamwater of 13 wildland watersheds managed under four management scenarios: (A) no grazing, (B) grazing without management, (C) grazing with management for livestock distribution, and (D) grazing with management for livestock distribution and with cultural practices to increase forage. Scenario D equated intensive grazing management to maximize livestock production, including practices to attain uniform livestock distribution and improve forage production with cultural practices such as seeding, fertilizing, and forest thinning.

The researchers found that FC levels in streams associated with scenario D were significantly higher than those of the other streams. Most of the A and C areas had FC levels less than 100 FC/L. Only one sample was available for scenario B and it was 150/L. FC levels for scenario D, on the other hand, ranged from 190/L to 2,270/L. A single sample from C was almost as high, 650/L. The higher elevations in these areas were attributed to the higher density of cattle in Strategy D areas (2.8 ha per animal unit month (AUM) compared to 8.2 and 7.7 ha/AUM for B and C. Also, vegetative characteristics played a role in that the areas with higher FC levels also had meadows desirable for grazing right beside the streams (Tiedemann et al, 1988).

Tiedemann et al (1988) also cited studies demonstrating that cattle noticeably increased fecal coliform counts. Some of these studies noted fecal coliform levels having up to a 10-fold increase over background levels (Coltharp and Darling, 1973; Doran and Linn, 1979; Gary et al., 1983; Skinner et al, 1974). In an earlier study, Tiedemann et al. (1987) found significant increases in streamwater FC counts with increased intensity of grazing management. The largest differences in FC concentrations (10X) occurred between control watersheds (no grazing) and watershed managed for maximum livestock production. Counts of FC in excess of 20000/L were observed when intensive management was used to maximize livestock production. These levels of FC can remain a problem even after the livestock is removed.

Stream Temperature Changes. Livestock can be extremely damaging to vegetation, as described earlier in this section. This disruption in vegetative cover can contribute to serious water quality degradation, especially if riparian areas are disrupted. In particular, vegetative damage (especially in riparian areas) can result in serious damage to aquatic habitats. Therefore, most of these impacts will be discussed in more detail in a later section of this document on aquatic ecosystems.

In terms of water quality, however, damage to vegetation can significantly alter a stream's temperature regime, leading to changes in fisheries and other aquatic life. Streamside vegetation is critical in terms of moderating stream temperatures. Because riparian vegetation intercepts and reduces the intensity of incoming solar radiation and reduces back-radiation, it serves as a form of insulator to the stream, preventing it from experiencing extreme temperatures or temperature ranges. Its shading effects in summer help to reduce excessive heating of the water. If the vegetation cover is decreased, summer stream temperatures can greatly increased, which contributes to a host of water quality problems, particularly a decrease in the amount of dissolved oxygen in the water. These changes to stream water quality may cause a shift in fish species, from salmonids to less sensitive species in many areas. By reducing the amount of back-radiation/reflection from the stream, vegetation also serves a moderating effect in winter. This also can enhance native fish survival, because if winter temperatures fall low enough, anchor ice can form on the bottom of the stream (Platts, 1991). The ability of plants to control stream temperatures depends on the size of the stream and the plant type. As a general rule, the larger the stream, the higher the streamside vegetation must be to effectively intercept the sun's rays over water (Platts, 1991).

Indirect Impacts on Terrestrial Ecosystems

Terrestrial Ecosystem Impacts. Most grazing studies examine changes in vegetation composition and the reduced range quality in terms of a loss of livestock carrying capacity. Little is known about impacts of sustained grazing on an ecosystem-wide level, particularly, impacts on wildlife. Dwyer et al. (1984) note that range management has focused on improvements to support increased livestock production, with little attention to maintaining plant and wildlife diversity within an ecosystem. Dwyer et al. (1984) cites both direct and indirect impacts on wildlife from livestock overgrazing. Direct impacts include competition for palatable species, while stress-producing modifications to the ecosystem induced by livestock (e.g. reduction in protective vegetation cover) are more indirect.

A consistent, direct impact of livestock overgrazing on rangeland is loss of vegetative diversity. Selective grazing by livestock tends to reduce the presence of palatable species while allowing a few, typically unpalatable and undesirable species to increase. The resulting change in plant composition lowers species diversity, changes species function, and reduces both the numbers and the variety of wildlife species the area can support (Dwyer, et al., 1984) To sustain a given wildlife population, the pre-grazing plant composition, structure and function within an ecosystem must remain in balance, following the introduction of livestock. Wildlife that depend on a limited number of plant species to provide a nutritionally optimal diet may be impacted as livestock can rapidly deplete limited food sources within a given area. The depletion of desirable vegetation species within an allotment forces wildlife into marginal, less desirable habitat and into eating less desirable/nutritious vegetation (GAO, 1991; Dwyer, et al., 1984).

Livestock impacts on rangelands extend beyond the direct loss of vegetation to modification of native habitat. Whole ecosystems may be impacted, and depending upon the fragility of the ecosystem, may be permanently altered. Some ecosystems are better able to withstand livestock and wildlife use; water sources, either in the form of precipitation or riparian zones, increase an ecosystem's ability to recover from stress. The increase of sagebrush and other bushy species in place of grasses is an indicator that fragile desert ecosystems have already been significantly impacted by overgrazing. The low rainfall, high temperatures, and high evaporation rates of these areas have produced plants and wildlife uniquely adapted to these regions. The adaptation of these ecosystems and their occupants to inherently harsh environments reduces their capacity to recover from disturbances, such as overgrazing (GAO, November 1991).

Over 250 native species are endangered, threatened or candidate species, in the southwestern Mojave, Sonoran, and Chihuahuan deserts. Poor management and/or overgrazing are factors identified as contributing to a decrease in preferred-diet plant species, destruction of habitat, and reduction of cover needed to hide from predators. In other cases, diseases may be transmitted from domestic to wild animals. In addition to their consumption of prime vegetation, poor management of livestock in the Sonoran desert have forced Sonoran pronghorn antelope away from traditional birthing grounds to less protected areas (GAO, November 1991).

Cosby (1978) noted that livestock grazing does not always impact wildlife negatively. Cosby observed several benefits of rotation grazing systems on wildlife when he found that deferring grazing in several units and altering the season of use actually increased vegetation diversity and cover. Cosby found sandhill cranes utilized grazed units regularly due to an increase in insect populations in the vicinity of "cowpattis". Similarly, native deer utilized units previously grazed to graze on new plant regrowth. Despite these findings, Cosby explains that this same scenario may not be feasible in a different region, and that all grazing treatments must be chosen carefully, on a site-specific basis.

Many livestock grazing researchers acknowledge the importance of avoiding grazing practices which result in the displacement of wildlife species, and to manage rangeland to maintain a healthy ecosystem complete with plant and wildlife diversity (Dwyer, et al., 1984; Carpenter, 1984). However, not all changes in species distribution, should be viewed as adverse impacts. The successional ecosystem stage (early, middle, or late) will help determine the appropriateness of maintaining species diversity and distribution as part of an overall range management plan.

Indirect Impacts on Aquatic Ecosystems

Effects of poor livestock and wildlife grazing management on stream hydromodification and water quality can have serious ramifications on aquatic ecosystems. Potential impacts such as bacterial contamination, increased sedimentation, and temperature changing can reduce the quality of the stream's ambient environment so as to affect the composition and health of aquatic organisms. Likewise, reduction of vegetation and increased runoff and flow may damage the stream's usefulness as aquatic habitat. Such impacts can originate from livestock and wildlife overgrazing in upland and riparian areas, although damage to riparian areas typically cause the most serious stresses to aquatic ecosystems. The following discussion focuses on overgrazing's adverse effects in riparian areas as these most closely and directly effect stream ecosystems. Also, much of the discussion will center on adverse effects on fish habitat; one important measure of the health of an aquatic ecosystem is by the nature and type of fish species present. The ability of an aquatic system to produce and support game fish is one way of measuring a healthy aquatic environment. For example, Van Velson (1979) found that rough fish comprised 88 percent of a fish population before relief from grazing and only 1 percent of the population after 8 years rest from grazing. Platts (1991) also examined a number of research studies, finding that in 20 of 21 studies, stream and riparian habitats were degraded by livestock grazing and that those habitats improved when grazing was eliminated. The majority of the studies also found reductions in salmonid fish populations related to the grazing-related habitat destruction.

Earlier sections of this document described how overgrazing of livestock and wildlife can affect the density and composition of vegetative cover. In upland areas, these impacts can lead to soil compaction and increased runoff. The hydrologic modifications to streams associated with increased runoff effectively destroys much of the desirable stream habitats.

As reported in Platts (1990), ideal trout spawning area is typically devoid of boulders, low in fine sediments, and high in gravel and small rubble. It also has a number of deep pools, well-aerated water, and ample cover and shade. Many of these necessary qualities of trout habitat can be wiped out by excess runoff and sedimentation. For example, increased flows can wipe out cover and habitat provided by fallen trees and brush.

Impacts of overgrazing on vegetation in riparian areas can affect aquatic ecosystems in a number of ways. Some of the impacts are similar to those associated with upland areas, but the impacts from damage to riparian areas are much more extensive and severe. Because of the proximity of riparian areas to streams, they are intimately connected to the stream ecosystem. Also, they are the preferred grazing ground of livestock and winter range for wildlife, thus concentrating much of the grazing-related damage to those areas. Livestock prefer to graze in riparian areas because they provide easily accessible water, favorable terrain, good cover, soft soil, a more favorable microclimate, and an abundant supply of lush palatable forage. Even though riparian areas represent a very small proportion of total rangeland, they provide much of the vegetation consumed by livestock because it is such a preferred grazing area. For example, Roath and Krueger (1982) reported that although the riparian zone constituted only 1.9 percent of the area on one allotment in Oregon's Blue Mountains, it produced 81 percent of the vegetation removed by cattle. Some of the ways that overgrazing (especially in riparian areas) can impact aquatic ecosystems are summarized below.

<u>Disruption/Reduction to Ecosystem Sources</u>. The riparian area serves as a source of energy to the aquatic ecosystem, by providing energy to streams in the form of dissolved organic compounds and particulate organic detritus. Benthic detritivores, the stream bottom bacteria, fungi and invertebrates that feed on the detritus, form the basis of the aquatic food chain. They pass on this energy when they are consumed in turn by larger benthic fauna and eventually by fish (U.S. Department of Agriculture, Forest Service, 1991). Riparian vegetation produces the bulk of the detritus that provides up to 90 percent of the organic matter necessary to support the headwater stream communities (Kauffman and Krueger, 1984). Platts (1991) stated that organic matter from riparian vegetation comprised roughly 50 percent of the stream's nutrient energy supply for the food chain. Disruption (i.e., change in cover density and composition) to riparian vegetation can severely reduce the extent of organic inputs to the stream, thus alter the energy of the ecosystem. Streamside

vegetation is also important to the production of fish food. It provides habitat for terrestrial insects which are important food for salmonids and other fish species.

Moderator of Stream Temperatures. Streamside vegetation is critical when it comes to moderating the temperature of streams. It shades the stream and therefore influences water temperature. A loss of vegetative cover can result in increased temperatures in summers, decreased temperatures in winter, and a greater daily range of temperatures at all times. Kauffman and Krueger (1984) reported on literature that showed damage to riparian areas caused increases in stream temperature (one study showed that maximum daily temperatures outside of a grazing enclosure averaged 7 degrees centigrade higher than those within the enclosure) and a greater range in temperature fluctuation (average daily fluctuation was 15 C outside of the enclosure and 7 C inside the enclosure). The increase in summer temperatures increases a trout's demand for dissolved oxygen, while at the same time, reduces the amount of dissolved oxygen in the water. This can cause a shift in fish species, from salmonids to nongame fish in many areas. Vegetation also serves a moderating effect in winter, which can enhance native fish survival. If winter temperatures fall low enough, anchor ice can form on the bottom of the stream. Streams with little or no vegetative canopy are very susceptible to the formation of anchor ice (Platts, 1991; U.S. Department of Agriculture, 1991).

Habitat Benefits. Riparian vegetation strongly influences the quality of habitat for anadromous and resident coldwater fish by providing shade, ameliorating in-stream temperature fluctuations, and providing cover (Kauffman and Krueger, 1984). Many studies have demonstrated the importance of cover to fish by showing that declines in salmonid abundance occur as stream cover is reduced and an increase in salmonid abundance as cover is added. The fringe of bordering riparian vegetation is essential for building and maintaining the stream structure necessary for productive aquatic habitats. This vegetation not only provides cover, but buffers the stream from incoming sediments and other pollutants and the effects of excessive flow (Platts, 1991). For one, fisheries habitat in streams is enhanced by the addition of large woody debris to the stream channel which forms pools and important rearing areas. This debris also provides cover from predators and protection from high flows. Large stable debris also provides the mechanism by which the detritus is held long enough to be processed by the invertebrate community. Without debris dams, much of the organic input from streamside vegetation would be washed downstream without contributing to the life processes of the aquatic food chain (U.S. Department of Agriculture, Forest Service, 1991). Each type of vegetation exerts a special function, as summarized in Platts (1991):

- Trees, shrubs, and sedges provide shade and streambank stability because of their large size and massive root systems. As trees mature and fall into or across streams, they create high quality pools and rifles. Their large mass also helps control the slope and stability of the channel. Input of this large organic debris is essential for maintaining stream stability. In many aquatic habitats, if it were not for this type of input, the channel would degrade and soon flow on bedrock, leaving insufficient spawning gravels and few high-quality rearing pools for fish.

- Brush also builds stability in stream banks through its root systems and litter fall.

- Grasses form the vegetative mats and sod banks that reduce surface erosion and mass wasting of stream banks.

Sediment Trapping. Riparian vegetation is important in slowing the overland flow of water and trapping sediment, therefore contributing to the building of bank form (Platts, 1990). Streamside vegetation is also important as it creates streambank stability. Vegetative mats reduce water velocity along the stream edge, causing sediments to settle out and become part of the bank. This helps to contribute nutrients to the bank soils and increases plant production and vigor. It also reduces the amount of sediments input to the stream (Platts, 1991).

In sum, by affecting the health and vigor of vegetation (especially riparian areas), poor grazing management practices can cause a number of problems that can damage aquatic ecosystems. These are briefly reiterated in the following bullets presented in Platts (1990). Reductions/loss in vegetation can:

- Increase average stream temperatures in summer, decrease them in winter, and expand daily temperature ranges.
- Reduce stream bank strength, enabling sedimentation and erosion, and reducing bank building through sediment deposition.
- Increase the erosive energy of water.
- Amplify effects of floods, ice, or debris flow, or animal trampling.
- Reduce water purification benefits that vegetation provides through infiltration and sediment removal.
- Reduce the ability of riparian areas to contribute to ground water recharge.
- Reduce flood control benefits.

POSSIBLE PREVENTION/MITIGATION MEASURES

This section identifies techniques that may be appropriate for mitigation of potential impacts caused by grazing activities. Mitigation should be evaluated on a site-specific basis and the following measures should only be used as a guide to measures that might be available should the reviewer determine they may be appropriate.

- Active management of livestock grazing allotments typically includes consideration of the following variables in different combinations : 1. grazing frequency, includes complete rest ; 2. livestock stocking rates; 3. livestock distribution; 4. season and timing of forage use; 5. livestock kind and class; 6. control of wildlife herd size and conflicts; 7. forage utilization; and 8. rehabilitation. Active management using these variables may increase forage, as well as improve habitat.
- Avoid high intensity, long duration grazing. The level of utilization must allow for regrowth of vegetation in order to maintain the productive capacity of the pasture.
- Encourage a greater level of control over the numbers of livestock and wildlife and time spent on each allotment.
- Encourage a greater level of oversight on allotments: more frequent assessment of utilization levels and quicker response to move livestock when utilization levels are attained may keep the area from being overgrazed.
- Separate riparian zone from other pastures and develop separate management plans, and if necessary, exclude livestock from riparian (or upland) areas until the desired level of recovery is attained.
- Fence or prevent direct access to streams in riparian areas to reduce trampling, damage of vegetation and the associated channel modification problems (may be costly to maintain, however).
- Use permanent exclosures in areas of high risk or extreme sensitivity where the likelihood of damage is high and the potential for restoration is low.
- Control livestock and wildlife grazing in areas predisposed to damage during periods of high sensitivity (adequate management plans).
- Use planned grazing systems to maintain plant vigor and desired species composition.
- Intensive practices (reseeding, weed control) may be necessary for extremely degraded pastures.
- Late season grazing should occur after the growth of warm season species has peaked and seeds have been produced.
- Know dynamics of plant species within an allotment and their capacity for regrowth.
- Evaluate type of livestock grazed and grazing intensity based on predicted impact to wildlife.

- Periodic minor ground shaping may be necessary to encourage dispersed flow and prevent concentrated flow.

- Plant compatible native trees or shrubs to reduce runoff, establish roots, and provide shade.

- Monitor progress of vegetation growth, bank and channel stability, and overall vitality of rangeland and riparian areas. Seasonal photographs may aid in this effort.

- Stabilize streambanks against erosion, although natural vegetative cover is preferred, artificial means of stabilization such as rubble, concrete or riprap may be necessary.

- Consider use of "in-stream" structures such as gabions, small rock dams, debris catchers, individual boulder placement, rock jetties, or silt log drops, to stabilize stream channels against excessive incision and/or widening.

- Plan periods of rest from grazing to stabilize streams.

- Consider changes in land use allocations, especially in or adjacent to degraded areas.

- Retain flexibility in allotment permits to account for special circumstances, such as excluding livestock during drought periods or other special circumstances, if necessary.

- Monitoring of rangelands is an important activity that will provide opportunity to identify and mitigate impacts. Conduct follow-up monitoring of range trends including conditions and utilizations. Alter actions based on monitoring data.

SUMMARY OF INFORMATION THAT SHOULD BE ADDRESSED IN NEPA DOCUMENTATION

The following is a list of questions that may be appropriate to ask about grazing when reviewing NEPA documentation.

What are the objectives of the management plan? Has a clear idea of the management plan objectives been presented?

Determine what factor, such as bank instability or loss of woody plants, is of primary concern.

Is the area suitable for grazing? Has the kind and class of livestock and the duration and intensity of livestock grazing best suited to the area been determined?

Has the document identified specific species (plant and animal) in the area, what sources were used to determine this, how does it compare with other information on the area?

Are utilization levels related to the specific species of vegetation present?

What utilization levels are planned for this allotment? What is the planned monitoring frequency for the allotment?

How will action be altered or modified based on monitoring information? What are the triggers for determining alterations?

Are there any endangered or threatened species in the area?

Has sufficient forage been allocated to wild herbivores in the riparian management plan? What is considered sufficient?

What tools (fencing, herding cattle/sheep regularly, duration) are proposed to effectively manage the allotment?

What is the seasonal distribution of the allotment (spring, summer have higher production than fall/spring)?

Are any special managements employed in riparian areas? How will stream areas be protected, especially stream banks?

What is the estimated impact on local groundwater, and how will this be monitored?

Have the potential cumulative impacts been described?

What are the designated beneficial uses of water bodies potentially affected by the grazing allotment?

Are these beneficial uses impaired due to exceedance of water quality standards? What is the cause of the impairment?

STATUTORY AND REGULATORY FRAMEWORK

In addition to the National Environmental Policy Act of 1969 (NEPA), there are specific statutes that provide Federal land managers with authority to allow and control grazing on Federal lands under their jurisdiction. Typically, each land managing agency has its own implementing regulations that correlate to each statute's authorities and requirements. In addition to these statutes, there are broad-reaching Federal statutes oriented toward environmental protection, such as the Clean Water Act, and the Federal Insecticide, Fungicide and Rodenticide Act, that may also apply to grazing operations on Federal lands. Explained briefly below are the statutes most appropriately described in the context of grazing.

Taylor Grazing Act. As discussed above, the system of free access to Federal lands ended with the passage of the Taylor Grazing Act in 1934. This was the first official Federal effort at livestock management and placed the administration of the public lands under the U.S. Grazing Service, later to become the BLM.

Multiple Use Sustained Yield Act of 1960. This statute promoted multiple-use management of national forest lands, not limiting the uses based solely on economic returns. The term "multiple-use" denotes management of the lands and their renewable resources in a combination of ways that would "best meet the needs of the American people."

Forest and Rangelands Renewable Resource Planning Act. Passed in 1974, four years after the Public Land Law Review Commission completed its broad review of Federal land policies, this act was an attempt to encourage better economic management of the national forests, as well as providing opportunity for public participation, timber sales, and reforestation.

National Forest Management Act. This statute, passed in 1976, continued an initiative to engage in land-use and resource planning. Like the Forest and Rangelands Renewable Resource Planning Act of 1974, NFMA emphasizes resource inventory, cost/benefit analysis, improvement of the environment, interdisciplinary planning, and public involvement (Clawson, 1983). Though this act encouraged high economic standards, some sections maintain constraints on attainment of full economic management of the federal lands and provided terms for carrying out a multiple-use/sustained yield policy. National grasslands were bought under Forest Service management through the Bankhead-Jones Farm Tenant Act.

Federal Land Policy and Management Act (FLPMA). Passed in 1976, this Statute serves as comprehensive multiple-use legislation for public lands managed by the BLM and supports the notion of public land retention to manage these lands on the basis of sustained yield. FLPMA is also a planning act endorsing multiple-use of resources. Basic principles of the FLPMA include land use planning with public participation, protection of the environment with the cost of damage supplied by the user, receipt of fair market price for private use of public resources, and cooperation with state and local officials. (Brubaker, 1984)

Public Rangelands Improvement Act. Congress passed this Act in 1978 intending to improve the condition of the nation's public rangelands, roughly 268 million acres, and alter the grazing fee formula on Federal lands. The Act prompted an increase in grazing fees from $1.51 per animal unit month (AUM) to $1.89 per AUM. In 1986, Executive Order 12548 extended use of the formula indefinitely. The Public Rangelands Improvement Act also directed the Departments of Agriculture and Interior to maintain an on-going inventory of range conditions, authorized additional funding for range improvement, and encouraged the development of improved allotment management plans.

<u>Clean Water Act</u>. Two main provisions within the Clean Water Act affect grazing activities. Both of these provisions primarily consider grazing as an activity that contributes to nonpoint source pollution; grazing is, therefore, addressed within the context of nonpoint source pollution programs and regulations, specifically, the following:

- Clean Water Act Section 319 - Nonpoint Source Program: This is the principal provision in the CWA that addresses nonpoint source pollution. The program provides Federal funding to qualifying states for the control of nonpoint sources of pollution. To be eligible for funding, States must develop an assessment report detailing the extent of nonpoint source pollution and a management program specifying nonpoint source programs and controls.

- Clean Water Act Section 320 - National Estuary Program: This program may affect grazing activities if such activities occur in one of the estuaries targeted for the program (e.g., Puget Sound, Galveston Bay). This program focuses on point and nonpoint source pollution. EPA assists state, regional, and local governments in developing comprehensive conservation and management plans that recommend corrective actions to restore estuarine water quality. Currently, the majority of the NEP targeted estuaries are located near fairly urbanized areas and issues associated with grazing on Federal lands are not likely to be a high priority.

- Coastal Zone Act Reauthorization Amendments (CZARA): A relatively new program, currently being developed jointly by EPA and NOAA, CZARA has great potential for promoting broad-based nonpoint source pollution controls (including approaches affecting grazing) in coastal areas. Specifically, section 6217 of CZARA requires that states with an approved coastal zone management program develop Coastal Nonpoint Pollution Control Programs to be approved by EPA and NOAA. The major emphasis of the CZARA program is to develop and implement "management measures" for nonpoint source control to restore and protect coastal waters. Management measures defined as economically achievable measures (e.g. best management practices, citing criteria, operating methods) that will control nonpoint source pollution to the greatest degree possible, are required for many different categories of nonpoint source pollution, including grazing.

  The management measure for grazing was developed as part of the agricultural component of the coastal nonpoint source program. The measure focuses on the protection of sensitive areas and the implementation of conservation management systems and/or activity plans. Figure 6 defines the grazing management measure in detail.

  Each CZARA defined management measure essentially represents a specific nonpoint source program goal. Although the States are given a great deal of flexibility in achieving the specified management measures, EPA provided extensive technical guidance (EPA, 1993) on practices that could be used to meet the management measure goals. In the area of grazing, EPA recommended some of the following practices:

- Grazing Management Systems (as defined by the SCS) - deferred grazing, planned grazing, proper grazing use, proper woodland grazing, pasture and hay land management;

- Alternate Water Supplies (as defined by the SCS) - pipelines, ponds, troughs or tanks, wells, spring development;
- Livestock Access Limitation (as defined by the SCS) - fencing, livestock exclusion, stabilized stream crossings;
- Vegetative Stabilization (as defined by the SCS) - pasture and hay land planting, range seeding, critical area planting, brush and weed management, prescribed burning.

The CZARA program provides another important approach to reducing the effects of overgrazing on the natural environment. Although CZARA currently only applies to coastal states, there is a chance that its scope may be expanded inland as part of the overall CWA Reauthorization Amendments.

Figure 6. CZARA Grazing Management Measure (EPA, 1993)

Protect range, pasture and other grazing lands:

(1) By implementing one or more of the following to protect sensitive areas (such as streambanks, wetlands, estuaries, ponds, lake shores, and riparian zones):

(a) Exclude livestock,
(b) Provide stream crossings or hardened watering access for drinking,
(c) Provide alternative drinking water locations,
(d) Locate salt and additional shade, if needed, away from sensitive areas, or
(e) Use improved grazing management (e.g., herding)

to reduce the physical disturbance and reduce direct loading of animal waste and sediment caused by livestock; and

(2) By achieving either of the following on all range, pasture, and other grazing lands not addressed under (1):

(a) Implement the range and pasture components of a Conservation Management System (CMS) as defined in the Field Office Technical Guide of progressive planning approach of the USDA-Soil Conservation Service (SCS) to reduce erosion, or
(b) Maintain range, pasture, and other grazing lands in accordance with activity plans established by either the Bureau of Land Management of the U.S. Department of the Interior or the Forest Service of USDA.

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OCHOCO WILD HORSE HERD MANAGEMENT PLAN OBJECTIONS

Melinda Kestler

Appendix C

Comparative Reproductive Biology of North American Feral Horses, Jay F. Kirkpatrick, PhD, and John W. Turner, Jr., PhD; Equine Veterinary Science, Volume 6, Number 5, pages 224-230, PDF.

Habitat Evaluation: Guidance for the Review of Environmental Impact Assessment Documents U.S. Environmental Protection Agency Pgs. 1-129, PDF.

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Reviewed

Special Issue:
The Feral Horse

COMPARATIVE REPRODUCTIVE BIOLOGY OF NORTH AMERICAN FERAL HORSES

Jay F. Kirkpatrick, PhD[1], and John W. Turner, Jr., PhD[2]

ABSTRACT

Recent studies have suggested that various aspects of reproductive biology are strikingly different among the many herds of feral horses and ponies (*E. caballus*) in North America. The greatest differences include (1) sharply seasonal versus year-round mating and foaling patterns, (2) mare behavior at the time of parturition, (3) forced copulation and incest, (4) exclusive breeding by a single harem stallion vs breeding by two or more stallions, and (5) fecundity. The causes for these differences are discussed in terms of genetic origins of the various herds, the length of time each herd has been in a free-roaming state and subject to the forces of natural selection, the ecology of the ranges inhabited by these horses, population density, and sex ratios.

There is no record of scientific studies of the North American feral horse (*Equus caballus*) prior to 1970, despite large numbers of the animals, particularly in the western U.S. In the early 1970's developing interest in the preservation of these horses led to protective legislation, and a new awareness of the feral horse situation stimulated numerous studies of feral horse biology. Since 1970 at least fourteen different feral horse herds have been studied with sufficient care and breadth to provide some understanding of the social organization, behavior, and reproductive biology of feral horses.

Attempts have been made to structure management policies around the available biological data, but a failure to view these data in a comparative context has largely obscured the diverse nature of the feral horse herds. This has led to confusion and disagreement among those who make policy. Similarly, the increase in interest in these horses has spawned a proliferation of popular literature, which, while interesting, is often without sound scientific foundation. Finally, the origin of most feral horse herds is unknown and speculative at best. There is little reliable data to document how long these animals have been in a free-roaming state and subject to the forces of natural selection.

Klingel[22] reviewed the comparative social organization of wild equids and in a later paper[21] made the first attempt to examine the comparative biology of North American feral equids. This latter review included data from six feral horse ranges, and it suggested that the feral horse herds of North America represented a wide spectrum of social organization, behavior and biology. McCort[23] reviewed the comparative behavior of North American feral horses on a more comprehensive level and also suggested that there were differences among the animals from different herds and ranges. Many aspects of the evolution and the present patterns of reproductive biology remain unaddressed. It is the purpose of this review to examine the comparative reproductive biology of feral horses from 14 different and genetically isolated herds in North America. The data were taken from 37 different studies. A summary of herd locations and references is given in Table 1. Specifically, this review will examine reproduction as a function of seasonality, age, and the social structure of the band, as well as fecundity, forced copulation, incest, and the behavior of mares during foaling.

Seasonality: The majority of studies indicate that feral horses are sharply seasonal with respect to breeding and foaling. Horses from the Pryor Mountain herd[9,10,18], Winnemucca (Hall, unpublished data), Stone Cabin Valley[12], Red Desert[4], Carson National Forest[29,20], Alberta[35], and horses on Assateague Island[16,17], all have a well defined breeding and foaling season. Among these horses breeding commences in March and ends in August, with peak activity in May and early June. Foaling after August 31 is very rare and is best characterized by the Assateague horses, where 13% of 86 foals born over an eight year period

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TABLE 1	
Herd Identification and Reference	
Herd Location	**Reference**
Pryor Mountain (Montana)	Feist, 1971; Hall, 1972 Perkins, *et al*, 1979; Turner, *et al*, 1979, 1981; Kirkpatrick & Turner, 1983; Feist & McCullough, 1975; Kirkpatrick, *et al*, 1977; Angle, *et al*, 1979
Western Alberta	Salter, 1978
Red Desert (Wyoming)	Miller, 1979, 1981, 1983 Miller & Denniston, 1979 Denniston, 1979; Boyd, 1979, 1980
Stone Cabin Valley (Nevada)	Green & Green, 1977
Wassuk Range (Nevada)	Pellegrini, 1971
Carson National Forest (New Mexico)	Nelson, 1978, 1980
Grand Canyon (Arizona)	Berger, 1977
Granite Range (Nevada)	Berger, 1983
Winnemucca (Nevada)	Hall (unpublished)
Assateague Island (Maryland)	Keiper, 1976, 1979; Keiper & Houpt, 1984; Zervanos & Keiper, 1979; Houpt & Keiper, 1984; NPS, 1985
Shakelford Island (North Carolina)	Rubenstein, 1981
Sable Island (Nova Scotia)	Welsh, 1975
Challis (Idaho)	Turner & Kirkpatrick, 1982 Kirkpatrick, et al, 1982 Seal & Plotka, 1983
Beaty's Butte (Oregon)	Eberhardt, *et al*, 1982

appeared in April, 52% in May, 22.6% in June, 10.4% in July, and less than 1.0% in August and September[17]. Salter [35] reported that 97.3% of foals were born by June 31, in Alberta. Seasonal foaling patterns for three different herds are illustrated in Figure 1.

There are, however, a few notable exceptions to this seasonal pattern. Although, as noted above, the Carson National Forest horses showed a seasonal foaling pattern consistent with other herds, Nelson[30] witnessed mating activity throughout the year. Berger[3] also witnessed year-round mating activity among horses in the Granite Range of Nevada. In these two cases foaling was seasonal, from March to August, despite the occurrence of year-round mating. Welsh[43] on the other hand, reported both breeding activity and foaling throughout the entire year among the Sable Island horses despite peak activity for both parameters in the late spring.

The seasonal breeding and foaling pattern seen among most herds is most likely a function of mare reproductive physiology. Kirkpatrick and Turner[18] examined the incidence of ovulation and behavioral estrus among Pryor Mountain mares of proven fertility. Estrus cycles with proven ovulation occurred exclusively from April to August, although anovulatory behavioral estrus did occur occasionally from August to November. Since Nelson[30] and Berger[3] witnessed fall and winter mating without subsequent foaling during these seasons, this activity was most likely a result of the anovulatory behavioral estrus described above.

Why the Sable Island horses should breed successfully year-round, in the face of strict seasonality elsewhere in North America— and particularly when one considers the severity of North Atlantic winters— deserves more intense research but remains unanswered at the moment. It is noteworthy that the sharp seasonality of feral horses is in contrast to a less seasonal picture in domestic horses, where the incidence of ovulation ranges from 60% in September to about 20% in December.[11,42,36,31] It is doubtful that nutritional differences account for the disparity in the reproductive seasonality between feral and domestic horses. Kirkpatrick and Turner[18] placed captive feral mares from the Pryor Mountain herd on a relatively high plane of nutrition but ovulation still failed to occur after August. Kirkpatrick and Turner[18] speculated that natural selection has resulted in

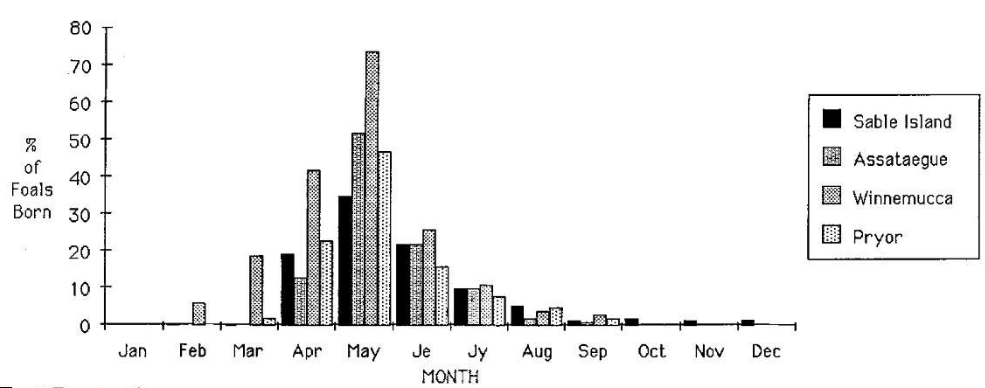

Figure 1. Seasonal Foal Production

a highly seasonal reproductive pattern which limits foaling to the period most favorable for survival of foals.

Sexual Maturity: Three years appears to be the age when foaling is first successful. In the Pryor Mountains mares did not foal until age three[13,10,33]. Among the Assateague horses, mares came into estrus during the second summer and mares younger than three never foaled[17]. In Alberta, Salter[35] commonly observed the breeding of two-year-old mares, but only one ever foaled as a result of these breedings. Welsh[43] noted essentially the same situation among mares of the Sable Island herd. The youngest sexually mature mare was two years old and the youngest successfully parturent mare was three. Boyd[4] also noted some sexually mature two-year-old mares breeding in the Red Desert, but no mares aged less than three ever foaled.

It also appears that three years is the critical age for successful reproduction among feral stallions in the Pryor Mountain herd[9,10,33,1] and the Sable Island horses[43]. However, Boyd[5] claimed that stallions became sexually mature at age two among Red Desert horses. Although Feist[9] reported occasional attempts by male foals in the Pryors to mount mares, as often as not the attempts to mount were made upon the mares' flanks, demonstrating the inexperience of the foal and bringing into question whether this behavior was actually sexual in nature.

Parturition: Normally the harem stallion will not permit mares to stray from the band, but an exception to this rule is seen at the time of parturition. At this time the mare will wander some distance from the band, usually to a secret or sheltered spot where she will foal. From one to three days later she will rejoin the band. This pattern has been reported in the Pryor Mountains,[9,10,13] the Wassuk Range,[32] Stone Cabin Valley,[12] Alberta[35] and Sable Island[43]. On Sable Island one mare was reported to use the exact same birth site for three years in a row.

The one exception to this pattern is seen among horses of the Red Desert[5] where mares seldom left their bands to foal, but rather simply laid down next to the band and gave birth. Boyd speculated that this behavior resulted from a lack of cover and simply the inability to find an isolated or hidden location. This seems unlikely, since there is considerable sage and greasewood on the Red Desert.

Forced Copulation and Incest: Few studies have directed any attention to these topics. Berger[3] reported that when new stallions assumed control of a harem in the Granite Range, it was common to see them harass mares already pregnant by the previous stallion. The harassment consisted of persistent and aggressive biting and chasing until abortion was presumably induced. Following abortion, the stallion forced copulation. The author, however, cited no evidence that pregnancy had been diagnosed in the mares studied, nor was any clear proof of abortion provided, thus these data must be viewed with caution. Welsh[43] reported witnessing a few forced matings among the Sable Island horses although he reported nothing comparable to the induced abortions described by Berger[3] nor were new harem stallions involved in the forced matings.

Hall[13] reported that stallions were never witnessed breeding their daughters in the Pryor Mountains. Welsh[43] however, reported a single incidence of a stallion breeding a daughter on Sable Island. In contrast, Keiper (personal communication) reported that 25% of the mares on Assateague Island remained with their fathers' bands and many were successfully bred.

Single vs Multiple Male Breeding: Perhaps no aspect of feral horse reproduction is more confusing or controversial than the issue of single male breeding versus multiple male breeding. McCort[23] has provided an excellent overview of the subject. In general, most breeding is carried out by a single sexually mature stallion and the pattern is remarkably consistent among horses of different herds. Herds in which the majority of bands have a single sexually mature stallion have been described in the Pryor Mountains,[9,13,33] the Wassuk Range,[32] Assateague Island,[15] the

Figure 2. The primary social unit in feral horses in the band, consisting of a harem stallion, mares and their offspring (left). Bands usually move independently, being intolerant of the close proximity of other bands. However, in some instances, especially when fleeing danger, several bands may move as a single group (right).

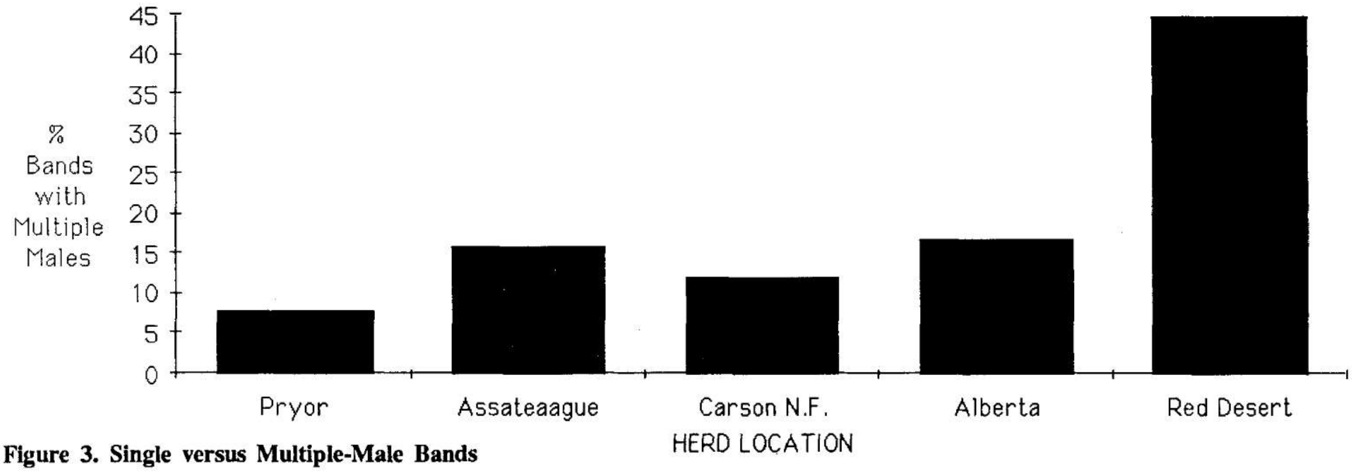

Figure 3. Single versus Multiple-Male Bands

Grand Canyon,[2] Carson National Forest,[30] Winnemucca, (Hall, unpublished data), Shakelford Island,[34] Alberta,[35] Sable Island,[43] and Challis[20].

Despite the predominance of single male bands in the Pryor Mountains, Hall[13] and Perkins *et al*,[33] reported a few instances where bands had two adult stallions. In these exceptional cases, however, the bands were usually large and always in excess of 10 horses. In 1971 Hall[13] gave the mean age of breeding stallions in the Pryors as 7.8 years while Perkins *et al*[33] described a range of 8-20 years with a mean of 12.8 (± 4.0 years) for Pryor harem stallions between 1974 and 1978. Of 17 bands described by Keiper[15] on Assateague Island, 12 possessed a single sexually mature stallion, four had more than one stallion, but most were sexually immature, and one herd had three young stallions of unknown age or maturity. Among the five multiple male bands there were no dominant stallions and they ruled by co-dominance. In 1985, the National Park Service[28] reported that Assateague Island held 107 horses, arranged in 14 bands, 12 of which were harem bands (two were bachelor bands). Among the 12 harem bands, only two had more than one stallion, and both were sons of the harem stallions. In Carson National Forest, Nelson[30] described 116 horses in 17 bands, fifteen of which had a single sexually mature stallion. Among the horses of Alberta, Salter[35] reported subordinate second males in only four of 23 bands.

A possible exception to the rule of single male bands was reported by Green and Green[12] at Stone Cabin Valley. Of 53 bands, 24 were described as having more than one stallion, but the investigators were careful to point out that there was no way of knowing the age of these animals or whether they were in fact sexually mature. The one clear exception to the single male rule is found amoung the horses of the Red Desert. Miller[24] reported that 23% to 45% of all bands in the Red Desert, observed between 1976 and 1979 were multiple male bands. These bands ranged in size from 3-17 horses, with a mean size of about 9.5, while single male bands ranged in size from 2-21 horses with a mean size of about six horses. The multiple male bands possessed 2-5 stallions with one always demonstrating dominance. In these multiple male bands, mating was accomplished 49% of the time by the dominant stallion, 42% of the time by the sub-dominant stallions, and 9% of the time by stallions from other bands. Equally unusual was the observation that in three instances mares in single male bands were bred by stallions from other bands, a condition not seen in any other herds. Also, 22 observations were reported of mares, in both single male and multiple male bands being mounted by more than one stallion.

Miller[24] also describes "feeding groups" of 50-150 horses within a one square mile area, in which individual bands cannot be distinguished from one another. This condition has not been witnessed in any other herd. Differences in single versus multiple-male bands, between herds can be visualized in Figure 3.

Why the Red Desert horses should present such a unique pattern among the continent's feral horses is a puzzle, however, Denniston[7] and Miller and Denniston[27] present three interesting theories for the development of multiple male bands. They first suggest that the herd is a loose aggregation of bands which represents a structured social unit. This would allow for some interaction between stallions of different bands, possibly resulting in a less rigid social structure. Thus it is possible that too much emphasis has been placed on the band as the primary social unit, and that subtle relationships between bands have gone unnoticed or unstudied.

These investigators also suggest that in some cases young males choose not to leave their bands and are not, for some reason, driven off by the band stallion. There is, however, little evidence to support this idea. A third possibility is the gradual loss of dominance by a band stallion, in a manner so slow and subtle that the interloper is never challenged. In this latter case a bachelor stallion will trail a band for some length of time and gradually join it, in degrees. This bachelor will follow closely, join in driving off other intruders, and finally join in mating. There may be a gradual increase in recognition of this bachelor by the band, and at the same time a gradual sense of acceptance by the band stallion. The fact remains, however interesting those theories, that the mating patterns of Red Desert horses are unique among the continent's feral equids.

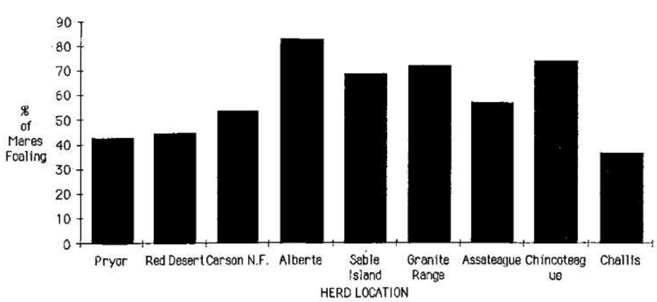

Figure 4. Fecundity by herd

A more probable cause for differences in the incidence of multiple male bands might be found in simple population density differences between herds, or the differences in sex ratios between different herds. Although these parameters are largely unstudied with respect to the issue of single versus multiple male bands, there are indications that they may profoundly affect band composition. Between 1979 and 1983, multiple male bands were extremely rare in the Challis, Idaho herd[39]. The incidence of multiple male bands has increased through 1985, perhaps as a result of round-ups which remove more mares than stallions, thereby increasing the competition for the position of band stallion.

Fecundity: Fecundity, measured by the index of foals/sexually mature mares must be viewed with caution. First, as has already been discussed, the age when mares reach sexual maturity may vary from herd to herd. Second, the reliability of age data provided by numerous investigators from widely disparate herds, using different methodologies, suggests room for a wide margin of error. Nevertheless, an examination of some available data indicates a very variable pattern from herd to herd.

Feist[9] and Feist and McCullough[10] reported that 43.2% of mature mares foaled during the 1970 season in the Pryor Mountain herd. In the Red Desert, Boyd[5] reported that 53% of mares three years or older foaled and 54.5% of mares four years or older foaled. Similar foaling rates were recorded at Stone Cabin Valley, 50%[12] and Carson National Forest, 53.8%[30]. In Alberta, however, Salter[35] reported a foaling rate of 83% and Welsh[43] reported an average foaling rate of 68.5% for the Sable Island horses.

In the Granite Range, Berger[3], made a distinction between fecundity in bands in which there was no change of band stallion (stable bands) and those in which band stallions had recently been replaced by new males (unstable bands). Among horses in unstable bands, Berger witnessed harassment of pregnant mares— which he claimed led to abortion - and 14 subsequent forced copulations with unreceptive mares. The foaling rate among the stable bands was 72 foals from 88 mares (81.8%) while the unstable bands produced only 9 foals from 24 mares (37.5%). Taken together, the Granite Range herd foaling rate was 72.3%.

Over an eight year period Keiper and Houpt[17] reported a 57.1% (±3.9%) foaling rate among the Assateague horses. The foaling rates were age-group specific and indicated that 23% of three-year-old mares foaled, 46% of four-year-olds, 53% of five-year-olds, and 69% of six-year-olds. On the adjoining Chincoteague refuge, just to the south of the Assateague range, the foaling rate was 74.4% (±2.4%).

Among mares in the Challis herd, Seal and Plotka[37] assessed pregnancy in 137 mares by measuring plasma progesterone, luteinizing hormone or pregnant mares serum gonadotropin, and estradiol-17 beta. Age specific pregnancy rates ranged from 35% among two-year-olds to 100% among 15-30 year-olds, with a collective 67.7% pregnancy rate for all mares age two to thirty. However, when foaling rates were determined a year later, among 35 mature mares in eight bands on the same range, Kirkpatrick *et al* [20] reported only 13 foals, or a rate of 37.1%.

Among Red Desert horses, Boyd[4,5] believes that consecutive year foaling is the rule, interrupted only by particularly severe winters. In 1978 she reported a foaling rate of 86 foals per 100 sexually mature horses (not just mares), but a year later, after a severe winter, the rate dropped to 54%. In support of this idea, Keiper[16] reported that only 10.2% of the Assateague ponies foaled on an alternate year basis, and Salter,[35] in Alberta, observed that among 12 mature mares with foals only 17% had not foaled the previous year. A similar pattern was described by Welsh[43] on Sable Island, where 26.7% of mature mares foaled three times in three years, 44.4% foaled two times/three years, and 8.9% did not foal once in three years. Welsh[43] also noted that 51% of parturient mares had conceived during foal heat and another 24% by the end of the next estrus. He interpreted this to mean that lactation does not suppress ovulation. Finally, the pregnancy rate study of Seal and Plotka[37] indicated that 52 of 85 mares had foals by their sides, or a 61.1% consecutive year pregnancy rate, although as has been pointed out earlier, it is unlikely that foaling rates were as high as pregnancy rates. These data suggest a significant embryonic loss over the winter months. Among horses from two different herds in Oregon, Eberhardt *et al*[8] reported 24.7 and 26.8 foals per 100 sexually mature adults, and an approximate 20% increase in herd size annually, between 1969 and 1980. Differences in foaling rates for different herds can be visualized in Figure 4.

One difference between foaling rates among horses of different herds can be explained by the physiological stress created by lactation and subsequent embryonic loss during severe winters. While no direct support for this hypothesis currently exists for feral horses, Keiper and Houpt[17] offer some strong circumstantial evidence. While the unmanaged horses of Assateague showed a 57.1% foaling rate, their counterparts on the Chincoteague refuge showed a 74.4% foaling rate. This latter herd has the foals weaned and removed in July of each year, removing the stress of lactation from the parent mares. Thus, among unmanaged herds it is logical to assume an inverse relationship between foal survival and foaling rates.

The major factor, however, in explaining differences in fecundity from herd to herd is most likely the differences in age structure between herds. The data from studies by Feist[9,10], Hall[13], Boyd[4,5], Welsh,[43] Keiper,[16] and Nelson[29,30] indicate that fecundity increases with the age-class of mares in the herd, thus, herds with older age structures will tend to have higher fecundity rates.

DISCUSSION

It is clear that there is variability among the reproductive parameters studied thus far in North American feral horses. At least five major forces may be assumed to influence reproduction to varying degrees. These forces include (1) the genetic origins of a given herd, (2) the ecology of the ranges inhabited by these animals, (3) population density, (4) age structure of the herd, and (5) sex ratios.

The genetic origins of the many herds are almost entirely unknown. The reintroduction of the horse to North America in 1519 and throughout the century which followed involved animals which have been described as Barbs and Andalusians, yet it remains pure speculation as to whether any of the contemporary herds are descendents of these horses. By the 1800's there were an estimated two million horses inhabiting North America[38]. Certainly the possibility that the original Barb and Andalusian genotypes may still exist, at least to some degree, cannot be dismissed. Empirical data also show modern domestic affinities exist in the contemporary feral horse herds. The largest infusion of these modern breeds undoubtedly occurred in the 1930's during the Great Depression, when thousands of horses were simply turned loose to fend for themselves.

At the present time only a single genetic study has been completed but certain genetic differences are obvious. Color patterns vary widely from herd to herd, as do average sizes and weights. The seasonal cyclic activity of feral mares, described here earlier, also varies greatly from breed to breed among domestic horses,[31] and points to the importance of genetics when examining biological differences between herds.

An alternative theory to significant genetic differences is epigenesis. Behavioral plasticity and adaptational flexibility are possible, without genetic change of populations. However, until comprehensive genetic studies are carried out, both theories have little substantive evidence to support them.

Habitat differences may have an important effect upon the reproductive biology of the feral horse. Availability of high quality grass, water sources, protective topography, size of the range (and therefore herd density), and weather patterns are but a few factors which may also dictate reproductive patterns and success. The best example here is the difference between foaling behavior of the Red Desert mares, where the mare simply foals next to the band, and that of mares from all other ranges, where they wander off to some hidden location to foal. Boyd[5] attributes this behavior to the open country of the Red Desert and the lack of suitable habitat in which to hide. Rubenstein[34] finds a correlation between geographical features of Shakelford Island— areas where the land was extremely narrow and visibility unrestricted— and the existence of territorial behavior.

In addressing the importance of the above forces in moderating reproduction, population density probably plays as large a role as any. The rodent studies of Christian[6] certainly support this view. In terms of horses it can logically be assumed that nutritional planes and population density will be inversely related and that reproductive success will depend upon adequate nutrition. In addition, the stability of band structure can be assumed to be inversely related to population density. Herd densities in turn may be a function of the age-class profile of the herd. A herd with larger numbers of older mares will have a higher fecundity rate and will increase at a faster rate than a herd with fewer older age-class mares.

Sex ratios may also alter band stability between herds even if overall density is the same. Larger numbers of sexually mature stallions would create increased competition for the position of band stallion, and the work of Berger[3] suggests that band instability leads to lower fecundity rates.

It is most logical, however, to examine the comparative reproductive biology of the continent's feral horses as a function of all five factors mentioned above. Genetics provides the foundation or starting point from which to examine any biological variation, and certainly reproductive biology. Horses with older genotypes might be assumed to take less time to reach equilibrium with their environment than those of a more modern lineage. Habitat and a multitude of other environmental conditions will in turn drive the process of natural selection and the more hostile the environment, the more dramatic the results of selection. Finally, fluctuations in population densities, age-class, and sex ratios, whether man-caused or natural, can bring about rapid changes which can impinge upon reproductive biology.

A few of the differences among the reproductive parameters discussed above-age at sexual maturity, and reproductive success— may represent nothing more than normal-curve variability that would be expected in any biological system, particularly since considerable variation occurs within a given herd of horses. Consequently it is difficult to place too much weight on the importance of genetic differences and natural selection. However, some variability between herds is so striking— parturition behavior, seasonality, forced copulation and incest, and single versus multiple male breeding— that normal-curve variability cannot explain the differences. This suggests a genetic influence and change through natural selection.

Information about the reproductive biology of North American feral horses has increased significantly since 1971, and it should be apparent from this review that considerable diversity exists from herd to herd. At least five factors must influence the reproductive biology of these animals, and the need exists for a model which integrates all relevant factors and seeks the correlations with reproductive parameters.

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HABITAT EVALUATION:
GUIDANCE FOR THE REVIEW OF
ENVIRONMENTAL IMPACT ASSESSMENT DOCUMENTS

EPA Contract No. 68-C0-0070
Work Assignments B-21, 1-12

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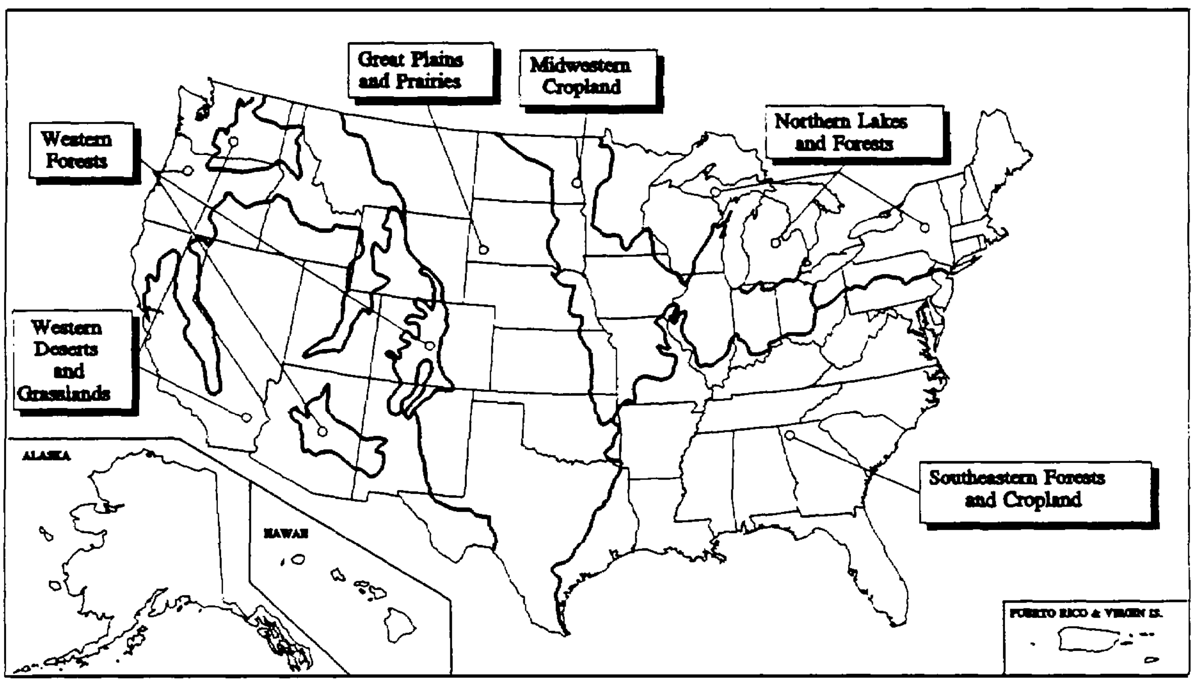
CONTENTS

INTRODUCTION

Based on the recent Science Advisory Board (SAB) report *Reducing Risk*, the U.S. Environmental Protection Agency (EPA) has concluded that habitat alteration and destruction are among the greatest risks to ecological and human welfare. The SAB specifically recommends that EPA consider reducing ecological risk to be as important as reducing human health risk. The recommendation states that EPA should protect ecosystems because they are essential to human health and a sustainable economy, and because they have intrinsic value.

This document is designed to assist NEPA reviewers in evaluating the ecological risks associated with the impacts of federal activities. The information provided will assist NEPA reviewers in developing informed comments for project scoping, EIS review, and section 309 analyses related to the issues of habitat loss and degradation. In particular, this document is designed to help reviewers recommend mitigations to prevent the loss of habitats. This document also should be useful to other EPA program offices and other federal agencies.

The first part of this document is a general discussion of habitat issues relevant to environmental analysis review; it should be read before the regional discussions. This section provides a basic description of habitat and its values, and of the degrading activities, impacts, and mitigations relevant to habitats in general. Eight *Regional Habitat Evaluation* sections, representing the six major habitat regions of the conterminous United States plus Alaska and Hawaii (see figure below), provide more specific



Habitat Regions of the United States

information on habitats of concern, values and trends, degrading activities and impacts, and potential mitigations. Specifically, each regional discussion includes a list of habitats of concern, a table of activities impacting habitats, and recommended mitigations for habitat conservation. Because each regional section considers only the major impacts affecting habitats in that region, reviewers should refer to different regions for discussions of other impacts that may be relevant to their specific project reviews. At the end of each section, basic guidelines are provided to aid in the environmental project reviewer's consideration of the full range of habitat impacts.

This document is not intended to serve as complete guidance or as a simplified checklist for environmental project review. In particular, this document focuses on activities occurring in the terrestrial environment, although impacts of these activities on wetlands and aquatic systems are also considered. Additional information on activities directly degrading aquatic systems should be reviewed where appropriate. It is expected that specific habitat issues relevant to the project site will be addressed, and that appropriate information on the ecology of the project site will be obtained. A list of useful institutional contacts is included with each regional discussion.

Habitat Conservation

Habitats are those environments or ecosystems that provide substantial ecological values and services such as fish and wildlife populations, nutrient cycling, water purification, and climate control. All natural areas contain definable units that can be called either ecosystems or habitats. In this document, the term habitat is equivalent to ecosystem and includes both the physical and biological components of the environment. All habitats are important for the conservation of ecological values at their specific location. However, certain habitats, and types of habitat, can be designated as "of special concern." For the purpose of this document, habitats of concern are defined as those sensitive environments whose degradation or loss results in significant diminution of ecosystem integrity or ecological values. The habitats of concern listed in this document represent the most obvious cases of loss of ecological values and services on a regional scale.

The following general discussion of habitat conservation begins with a summary of the important issues and steps involved in assessing habitats, follows with a working definition of habitats of concern, and continues with discussions of the values and services provided by habitats, the activities affecting habitats, the types of impacts caused by these activities, and potential mitigation measures to address these impacts on habitats.

Habitat Evaluation Methodology

The definition of habitat in this document is based on ecosystem values and functions. Therefore, it is necessary to present habitats as classes of similar ecosystems that contain a known set of ecological values and functions. The habitats discussed in this report are broad vegetation-based categories that include a range of more specific ecosystem types. While this document will categorize habitats and identify individual impacts, it must be remembered that each habitat is unique. An individual habitat must be evaluated in the context of its specific geographic location to determine its true value. At the same time, the effect of alterations to a habitat by degrading activities must be considered in terms of the impact on the entire landscape. Therefore, an ecological perspective is essential for the adequate consideration of habitat issues. This approach requires that the interactions of ecological components be considered, and that the unique characteristics of each ecosystem be evaluated. The following considerations should be central to any process of habitat evaluation:

- Apply an **ecosystem-level perspective** that considers the full range of interactions among habitat components.

- Assess the **cumulative effects** that arise from the additive and synergistic impacts of several degrading activities occurring over time or space.

- Analyze the true **effectiveness of mitigation** measures in conserving natural habitats and their ecological values.

It is common for habitat considerations to be neglected within environmental analysis because of the difficulties of individual site-specific assessments. To better address the consideration of impacts to habitat in environmental analyses, regional information on the impacts to habitats of concern and their mitigation can be used. Therefore, the sections that follow describe general habitats that are threatened with loss or degradation from human activities. The condition of these habitats, the activities that affect them, and potential mitigations for the impacts that degrade them are discussed.

The application of this regional information should improve the quality of environmental analyses of all kinds. Along with an ecosystem perspective, attention to cumulative effects, and measures of mitigation effectiveness, the following steps can be used to incorporate landscape-scale considerations into both regional-level and site-level environmental analyses:

Step 1. Review the status and trends of habitats in the regions under consideration.

Step 2. Identify habitats of concern for the region that may occur at the site.

Step 3. Analyze the impacts of all activities on the functions and values of these habitats.

Step 4. Derive mitigation measures to eliminate or ameliorate the impacts on habitats of concern.

Habitats of Concern

Virtually all of the natural environments in the United States have been degraded to some extent by the impacts of human activities. Even relatively pristine ecosystems are affected by the loss of contiguous habitats and other changes to the landscape. Therefore, the most important criterion for designation of a natural area as a priority concern is the importance of a habitat to the ecological integrity (i.e., the health and natural functioning) of the larger landscape or eco-complex (*sensu* Polunin and Worthington 1990). In this way, a habitat may be thought of as analogous to a "keystone species" within a biotic community. For practical reasons, rarity is often the criterion by which a habitat's value is determined. However, in assessing the value of a habitat, rarity, ecological functioning, regional diversity, and other important attributes also should be considered.

Definition of Habitat

The standard definition of *habitat* is based in the environment of individual species; for example,

> "Habitat is the environmental setting in which an animal or plant normally lives, grows, and reproduces" (NRC 1982);
>
> and
>
> "Habitat is the area which provides direct support for a given species, population, or community. It includes all environmental features that comprise an area such as air quality, water quality, vegetation and soil characteristics and water supply (including both surface and ground water)" (Fish and Wildlife Service, FR 46(15):7662-7663).

Although this definition has been important to the management and preservation of many individual species, it is inadequate for regional or global biodiversity protection efforts. Indeed, national inventories of species-specific habitat are not practical for most species, and in fact have been accomplished only for the critical habitats of endangered species (Flather and Hoekstra 1989). The need to address the conditions of a wide range of species, and biological diversity in general, requires an ecosystem approach to habitat inventory. For the purpose of this document, the following definition is used:

> **Habitat - a natural environment composed of both living organisms and physical components that function together as an ecological unit.**

In many contexts, this definition is synonymous with *ecosystem* or *sensitive environment*. It assumes that the natural condition of an environment is preferred because it represents a system that through evolution is most likely to provide the desired values of biological diversity and ecosystem functioning. Although the difficulties in classifying habitats or ecosystems have prevented the completion of adequate national inventories, different classifications have been used for specific purposes or for restricted locations. The National Wetlands Inventory of the U.S. Fish and Wildlife Service uses the widely accepted Cowardin classification system for wetlands and deepwater habitats (Cowardin et al. 1979). The U.S. Forest Service has used a variety of classification systems including the Forest and Range Environmental System (FRES) (Garrison et al. 1977) based on Küchler Potential Natural Vegetation units (1964) and Bailey Ecoregions (1976). The U.S. EPA has recently defined general classes of ecological resources for all habitat types as part of its Environmental Monitoring and Assessment Program (EMAP) (Hunsaker and Carpenter 1990). Greater resolution in habitat classification has been obtained by state natural heritage programs in coordination with The Nature Conservancy. Extensive natural heritage databases that once consisted of only species element occurrences now include "community" elements. At present, each state has a community classification, and many are working toward regional classifications. If this is accomplished, there will someday be national coverage of community types from which to base a quantitative assessment of habitats (Larry Master, personal communication).

Given the mixture of classification systems, systematic status and trends information is not available for most habitats (Southerland and Hirsch 1989). However, considerable information on the status and trends of individual species is available and can be useful in characterizing habitat status and trends. In fact, the Fish and Wildlife Service (FR 46(15):7662-7663) has developed the concept of evaluation species upon which they base analyses of environmental impact. The evaluation species include species of high public interest and economic value, and species that provide broad ecological representation. Environmental analyses can use identification of such "species of concern" as a useful starting point for identifying habitats of concern. Throughout this document, species status and trends will be included to the extent they reflect habitat conditions, but it must be remembered that they represent only a few of the many species in each habitat, all of which are required to maintain a healthy ecosystem and a full range of values and services.

General Habitat Types

Before colonization by Europeans, North America was covered from the Atlantic Ocean to west of the Mississippi River with diverse eastern deciduous forests of large oak, chestnut, beech, and maple; farther west spread a lush tallgrass prairie; beyond that was a semi-arid shortgrass prairie with regional deserts, grasslands, and coniferous forests (Norse 1990b). A nearly unlimited number of unique habitats existed within these regions, varying with soil conditions and topographic differences. The exploitation and manipulation of land by human activities has since eliminated or modified many of these habitats. This document uses the major land types of *forests*, *rangelands*, and *wetlands* to facilitate the identification of more specific habitats of concern. It focuses on habitat types that are repeated across the region and does not consider individual plant communities that vary with exact geographic location. The scale of these habitat types varies, and although a medium scale is applied in this document, it is important to remember that the following additional classes of habitats of concern should be considered in individual environmental analyses:

- Individual plant communities (e.g., those compiled by state natural heritage programs).
- Transitional habitats and functional mosaics of habitat, e.g., the sandhill-scrub-lake complex of the natural upland hardwood forest of Florida (Noss 1987).
- Landscape-scale ecosystems, or eco-complexes (e.g., the Chesapeake Bay watershed).

Values and Services of Habitats

Habitats provide the full complement of ecological values and services contained in a naturally evolved ecosystem. These include many services that have economic benefits, as well as aesthetic and moral values. All individual species values, overall biodiversity values, and ecosystem services are encompassed in ecological integrity. Therefore, it should be the objective of habitat conservation efforts to preserve the ecological integrity of habitats.

Species Values

Individual species are the values most often associated with habitats. Historically, commercial timber species and crop plants, and game animals and sport fish have been the most prized species; subsequently, noncommercial plants, nongame birds, endangered species, and other popular species have received attention. Those interested in species preservation are now viewing habitat conservation as a means of protecting species "wholesale" (Waller 1991). This is in contrast to the single-species approach required by the Endangered Species Act, often referred to as "emergency rescue operations" (as in the cases of the California condor and black-footed ferret). The best example of the habitat-based approach is the effort of The Nature Conservancy which has adopted a "coarse filter" approach to protecting species based on protecting the natural communities in which they reside. This approach provides protection for the majority of species, including unknown and undescribed ones.

The most visible values of any habitat are the many plant species that make it up. Plants are prized for their intrinsic value and for their roles in ecosystem functioning. Recently, previously ignored species are receiving attention for their contributions to genetic diversity. There is also ample evidence of the importance of habitat to animal populations. Among state wildlife and fish management agencies, habitat loss ranked first in national priority for all species, for big game, for small game, and for waterfowl (Flather and Hoekstra 1989). Habitat also ranked second to barriers to migration in importance for sustaining anadromous fish populations. Wildlife management efforts have had their greatest success with species (big game and some endangered species) for which habitat is abundant. Species whose habitat is declining in amount and quality are currently, and will continue to be, most threatened with extirpation (Thomas 1990).

Biological Diversity

The interest in preserving particular species has broadened in recent years to encompass a concern for all biotic resources under the general term "biological diversity." The Office of Technology Assessment (1987) defines biological diversity as

> "The variety and variability among living organisms and the ecological complexes in which they occur";

while the Keystone Dialogue on Biodiversity on Federal Lands (1990) defines biological diversity as

> "The variety of life and its processes."

Both of these definitions emphasize that biological diversity, or biodiversity, entails all ecosystem components and includes the myriad functions and values provided by the living organisms in each habitat. The number and relative frequency of items that make up biological diversity may be organized along the continuum from genes to species to ecosystems. The overall amount of genetic diversity is decreased when species diversity is lowered, as is species diversity when ecosystem diversity is lowered. For this reason, habitat loss and ecosystem degradation are the principal causes of reductions in biological diversity. Essentially, the conservation of habitat is the conservation of the ecological complexes that constitute biological diversity. In addition, the preservation of biological diversity may be the best means of protecting overall biological integrity and ecological health. Preserving biodiversity means maintaining the integrity of the genetic structure within populations, the richness of species within ecosystems, and the mosaic of ecosystems within the landscape (Norse 1990b).

Ecosystem Services

Although the conservation of individual species and overall biodiversity are essential to maintaining the ecological integrity of a habitat, a wide range of ecosystem functions must also be protected. Using a broad definition, habitat, like the ecosystem, is characterized by a particular energy flow, nutrient cycling, and capacity for self-perpetuation (given radiant energy from the sun). The services that ecosystems perform include serving as a store or sink for energy or materials, providing a pathway for nutrient transport, acting as a buffer against chemical changes, and producing the natural resources people use such as minerals, wood, food, water, and air (Hollis et al. 1988). A comprehensive list of ecosystem values is shown in the accompanying box.

Ecosystem Values[1]

Climate control:
- CO_2 sequestration
- evapotranspiration
- shading

Geomorphological control:
- wave and wind buffering
- erosion control
- sediment trapping
- soil building

Water supply:
- groundwater recharge
- floodflow alteration
- water supply

Energy and nutrient exchange:
- energy fixation
- carbon uptake
- nutrient uptake

Purification of resources:
- sediment/toxicant retention
- nutrient removal/transformation
- pollutant detoxification

Biotic resources:
- biotic productivity and food chain support
- genetic conservation of biodiversity
- fisheries
- wildlife diversity/abundance
- aquatic diversity/abundance
- aesthetics/cultural heritage

[1]Adapted from Race and Christie (1982), Adamus et al. (1987), Hinckley (1990), and Nash (1991).

Activities Impacting Habitats

After identifying the habitats of concern, the next important step is the linking of these habitats to the activities that cause their degradation or loss. The following major activities may cause the degradation or loss of habitats:

- Land conversion to industrial and residential land use.
- Land conversion to agriculture.
- Land conversion to transportation.
- Timber harvesting practices.
- Grazing practices.
- Mining practices.
- Water management practices.
- Military, recreational, and other activities.

Environmental analyses of these activities arise during both broad programmatic reviews and specific project environmental impact statements. The following common projects entail significant impacts to habitats and may require federal review:

- Community and public land use development, including planning, regulation, and federal funding for building construction and highway development.

- Renewable resource use and development (logging and grazing) on public lands or requiring permits.

- Energy production, including petroleum, natural gas, and coal development, extraction, generation, transmission, and use.

- Non-energy mineral resource development, processing, management, transport, and use.

- Water projects and permits for wetland modification.

- Natural resources conservation, including protection of environmentally critical areas.

This document focuses on the direct physical effects of the aforementioned activities on habitat extent and quality. However, another important source of impacts on habitat is the contamination of ecosystems from the pollution of the air, water, and land. Habitat pollution is addressed, in part, by the air quality, water quality, and hazardous substances programs of federal and state regulatory agencies. Therefore, habitat impacts from the generation of toxic and waste materials from manufacturing processes and fossil fuel combustion are not specifically addressed in this document. The following are examples of activities contributing to the contamination of habitats that should be added to the considerations in this document when a complete environmental analysis is prepared:

- Industrial and municipal discharges into water (e.g., toxic chemicals and conventional pollutants) and emissions into the air (e.g., acid deposition, gaseous phytotoxicants such as ozone, and global ozone depleting and greenhouse gases).

- Industrial and municipal·waste dumps and landfills (e.g., asbestos and plastics in the marine environment).
- Agricultural contamination (e.g., pesticide spraying and nutrient discharges from cultivated fields and livestock feedlots).
- Mining waste discharges (e.g., mercury, arsenic, cyanide, crude oil, drilling muds, and saline-produced waters).
- Military accidental releases (e.g., nerve gas and plutonium).

The following sections briefly discuss the history and impacts of the major activities on habitats.

Land Conversion

The conversion from one land use to another is the activity most severely affecting terrestrial environments. The type of land conversion depends on the end use of the land. In each case, the original natural characteristics of the land are eliminated, and the associated ecological values are modified to varying degrees. Urban conversions, as well as other large industrial and commercial development projects, severely alter natural conditions, seriously disrupting ecosystem functions and eliminating most ecological values. Residential development in suburban and rural areas usually maintains some plant and wildlife values while disrupting the natural ecosystem processes of the area. Similarly, conversion to traditional agriculture alters the natural vegetation and ecological processes while still providing some hedgerow areas for wildlife populations. Large-patch industrialized agriculture, however, usually removes all wildlife habitat. Conversions to industrial, residential, and agricultural uses occur on many scales, but often cover very large areas. In contrast, conversions to highways, railways, and power lines affect terrestrial environments more by fragmentation than by total area converted. Landfills and the development of recreational areas are other kinds of land use conversions, but ones that cover relatively small areas.

Land Conversion to Industrial and Residential Land Uses

Conversion of natural environments to industrial, commercial, and residential land use continues to increase with population and with the general suburbanization of many previously natural areas. The large urban areas of the east and west coasts continue to grow, reducing the natural areas in the corridors between them. Land conversion due to infrastructure construction and landfills also contributes to the development pressure on natural areas near urban centers. Urban growth is most rapid in the Sun Belt states.

Urban and suburban conversion of terrestrial environments is also occurring throughout the country as "spinoff development" following new road construction. Even in areas of relatively little or no overall population growth (such as the Northeast), spinoff development is a major cause of forest fragmentation and the decline of wildlife and bird populations. This effect is augmented by the increasing frequency of second home development in previously undeveloped regions.

Arid environments in the Southwest are rapidly being converted to urban and residential uses as a result of population growth. The Southern California region is a classic example of suburban sprawl where roadways, residential communities, and commercial development have expanded into previously pristine environments. Many underappreciated desert habitats are at risk because of this continued land conversion. Riparian areas are another environment at risk in the West from land conversion to industrial and residential development. Because of their proximity to water and their desirability for industrial and residential use, riparian areas are being disproportionately destroyed. Also because of their proximity to water, riparian areas are critical for many migratory bird and wildlife species.

Land Conversion to Agricultural Uses

The United States uses a large part of its available land area for livestock and crop production, an area totaling more than 900 million ac (U.S. EPA 1989). Over 400 million of these ac. are classified as cropland. More than 50% of this area is in the corn and wheat growing regions of the Midwest Cropland and Great Plains and Prairies Habitat Regions. Land conversion to agriculture has stabilized in recent years, and much of the conversion to urban uses is now occurring on old agricultural lands. Conversion to agriculture continues to be a regional problem depending on the pricing variability of specific crops. For example, bottomland hardwoods in the South have recently suffered from extensive conversion to soybeans.

Although total agricultural acreages are not changing, many important wildlife habitats are being lost as a result of large-patch agriculture, which causes the elimination of fence rows and ditch banks. Current agricultural practices, and certain "conservation" programs, provide incentives for cultivating previously uneconomical areas. For example, the construction of grass waterways in riparian areas is destroying wildlife habitat rather than conserving it.

The loss of riparian and bottomland hardwoods to agriculture in the Southeast represents one of the most significant losses of ecological values of terrestrial environments. Similarly, the conversion of wetlands and adjacent grasslands in the central and western United States is another impact that has had serious consequences for ecological values, in particular waterfowl populations in the Prairie Pothole Region and along the Pacific and Mississippi Flyways.

Land Conversion to Transportation Uses

Construction of highways, railways, and power line right-of-ways contributes to the degradation of terrestrial habitats, especially in less developed areas. Although the actual areas converted are small (27 million ac) the fragmentation of habitats is often severe (Frey and Hexem 1985). Powerlines and other transportation routes can be described as "disturbance corridors" that disrupt the natural, more homogeneous landscape (Barrett and Bohlen 1991). In forested environments, these disturbances cause (1) dramatic physical disruption to the continuous vegetative community; (2) disruption to the structure and function of wildlife habitat; and (3) impacts to resident wildlife, which must negotiate, tolerate, and cope with the habitat barriers. In addition, disturbance corridors created by forest fragmentation provide habitat for early successional plant and animal species. They replace forest trees with grasses and shrubs so that forest-interior species cannot nest. While they provide dispersal routes for small mammals such chipmunk and white-footed mice, they present barriers to many species.

The impacts of highway construction also represent an important problem in cumulative impact assessment. Although individual road segments are usually evaluated for potential environmental impact, it is actually the combined effect of the entire highway system that most seriously degrades terrestrial and wetland environments. In addition, the cumulative impact of several highway systems can seriously disrupt migratory pathways. As mentioned above, the building of roads is invariably accompanied by additional land conversions to industrial or residential use.

Both forested and nonforested environments can be disrupted by fragmentation due to highway construction. However, the dense canopy structure of certain shrublands may be most severely impacted by fragmentation. An example is the fragmenting of pocosin wetlands and uplands in the Southeast. Because of the scale at which many pocosin inhabitants move, highway development can effectively isolate much of the pocosin fauna.

Timber Harvesting

Since the early 1600s, 20 to 40% of the nation's original forest cover has been converted to other land uses, and much of what remains has been substantially altered as a result of past logging. Regeneration of timbered areas is increasing forest acreages in the East, but these numbers are more than offset by timber harvests in the West. Many of the remaining forests of the United States are being altered by timber harvesting practices that fragment, simplify, and degrade natural forests. The combination of clear cut logging and road building increases forest fragmentation and soil erosion. The clear-cut natural stands are often replaced with fewer and different tree species resulting in the loss of old-growth trees and natural forest habitats essential to a wide variety of wildlife.

Forest habitats are the forum for the most acute biodiversity issues facing the nation, including (1) decreases in contiguous old-growth forest that support the spotted owl in the Northwest, (2) the loss of old pines needed by the red-cockaded woodpeckers in the Southeast, (3) increased habitat fragmentation and forest edge causing declines in forest-interior songbirds, and (4) increasing ungulate populations in the East and Midwest (Waller 1991). These problems are primarily the result of clear-cut logging and the institution of short-rotation single-species plantations. All timber harvesting activities affect forests in two ways (Cutter et al. 1991):

- Like natural fires, timber harvesting allows sunlight to reach the ground and stimulate new growth, while the slash (limbs too small to use) contributes to increased nutrient release. Thus, like fire, harvesting is a catastrophic but temporary disruption that removes large amounts of soil, nutrients, and biomass from the ecosystem, changes water yields, and increases stream temperatures.

- Unlike natural disturbances, timber harvesting involves road building and the use of heavy equipment on the land; this causes damage and compaction to the soil surface and accelerates soil erosion beyond the rates following fires. Especially along steep slopes, surface erosion and landsliding produce heavy sediment loads to streams, degrading aquatic habitats and damaging fish and invertebrate populations; the loss of biomass in the form of logs slows reestablishment of new growth, and the lack of fire may retard regrowth from fire-adapted seeds.

The major impacts of timber harvesting on forest degradation and loss include four major problem areas that can be addressed on a national or regional basis:

- Loss of old-growth forests.
- Effect on critical ecosystems (such as Greater Yellowstone).
- Decrease in roadless areas or wildlands.
- Impacts of silvicultural practices (such as clear cutting).

Grazing

Widespread devastation of rangelands resulted from uncontrolled overgrazing between 1880 and 1935, and the damage was amplified by the drought years of the 1930s (Branson 1985). The enactment of the Taylor Grazing Act of 1934 reduced grazing pressure at that time. With the advancement of range management science and the moist years following 1960, range vegetation improved considerably. However, the U.S. Forest Service (1989) reports that 21% of its rangelands were still in "unsatisfactory" condition. The Bureau of Land Management (1989) reports that BLM rangeland condition is 33% good or better, 38% fair, and 13% poor.

Although the total area of rangeland has remained relatively constant, the condition of the range ecosystems has varied considerably with competition by livestock for forage and other factors. Cattle, sheep, and wild horses and burros have contributed to reduced forage and to changes in vegetation composition on the majority of U.S. rangelands. Grazing and fire suppression have allowed brush species to replace many of the grass forage species on 200 million ac of the Southwest (National Association of Conservation Districts 1979). As with forest habitats, the fragmentation of rangeland vegetation can negatively affect native fauna and ecosystem health.

Unfortunately, traditional rangeland improvement measures often run counter to wildlife conservation. Herbicides reduce vegetation diversity, as do practices that till under sites and convert vegetation to nonnative species, usually replacing pinyon juniper with exotic grasses. Management of brush invasion in the southwestern deserts, savannas, and southern Great Plains is perhaps the greatest problem affecting rangeland wildlife. While deer and turkey populations have increased, native range forage is reduced by the invasion of mesquite, juniper, cacti, acacia, sand sagebrush, creosote bush, tarbush, whitebrush, yucca, and others. Mechanical or chemical reduction of these scrubs, as well as sagebrush in the Northern Plains, decreases forage for many species including prairie chicken, sage grouse, quail, and pronghorn.

Grazing is also detrimental to hardwood forests, riparian habitats, and areas where livestock compact root systems or increase erosion. In general, grazing reduces structural diversity of forest understory (by eliminating plants, altering species composition, modifying growth form, and shifting seral stages) and can negatively impact forest bird communities. Of particular concern are the impacts of grazing on forested riparian zones, which support the majority of species in the rangeland environment.

Mining

Millions of hectares of marginal and barren land can be found in the United States, much of it due to mining activity. These areas are a source of acid mine drainage, surface runoff, erosion, and sedimentation, which create water pollution and land degradation problems. Mining activities leave a harsh environment for vegetation because of the lack of nutrients and organic matter, low pH, low water-retaining capacity, toxic levels of trace metals, compaction, and poor physical conditions of spoil material (Sopper 1988).

It is important to note that mining often occurs on the mountain-plain ecotone, an area of special importance to wildlife. Nonetheless, mining disturbs relatively less land area than other activities affecting terrestrial environments. Only 5.7 million ac were disturbed between 1930 and 1980 by surface mine excavation, subsidence from underground workings, and disposal of mining wastes. Additional areas have been impacted by haulroads, reservoirs, and railroads and highways to mining properties. Stream habitats have been affected by acid drainage and sedimentation. The greatest potential for increased mining impacts exists in the area of exploration and extraction of fossil fuels.

Nearly half of all U.S. land used for mining is concentrated in the states of Pennsylvania, Kentucky, West Virginia (2% of each state) or in Ohio, Illinois, and Indiana (1% of each state). California and Florida have also mined more than 250,000 ac (Johnson and Paone 1982). Intense mining also occurs in the Arizona copper region and the northern Minnesota's Mesabi Iron Range. Among federal lands, 732 million ac are available for leasing to surface and subsurface mineral development, the majority in the west and Alaska; currently, 95 million ac are leased to oil and gas, 2 million to geothermal, and 1.3 million to coal (USDA Forest Service 1989).

Mining impacts are substantial but variable depending on the mining method, the mineral, the processing technology, and the ecological nature of the site. Impacts include destruction or impairment of fragile ecosystems and wildlife habitats, contamination of surface and subsurface water supplies and soils from toxic chemicals and radioactivity, and adverse effects on scenic values.

Water Management

Damming activities, impoundments, and water diversions for municipalities, industry, and agriculture severely affect the natural water supply, resulting in the destruction of terrestrial, wetland, and aquatic environments. In particular, the reduction of streamflow from diversions of water for other uses adversely affects riparian habitats in the Southwest. The Corps of Engineers stream channelization projects affect large areas of both terrestrial and aquatic environments. In fact, few streams or waterways still run free to the ocean without diversion or management that affects their natural flow. The inundation of large areas for flood control and water supply has decreased in recent years, but still constitutes a major impact on local environments. In the Mississippi Basin (mid-south Alabama, Tennessee, eastern Texas, and Oklahoma), considerable acreage of bottomland hardwoods was lost to reservoir development between 1962 and 1985 (Gosselink and Lee 1987).

Changes in water quality, flow, and dam passage affect the success of anadromous fish populations, including recreationally important game species. In addition to the intrinsic value of these species, the degradation of important aquatic resources has a detrimental effect on many terrestrial systems, including migratory birds and riparian forests. The importance of wildlife impacts from hydropower activities is evidenced by the provisions for wildlife habitat mitigation in the Columbia Basin under the Northwest Power Act (Brown 1988).

Recreational, Military, and Other Activities

Several other human activities can seriously affect terrestrial environments. The introduction of nonnative species into wild areas also has the potential for devastating alterations of terrestrial habitats. Even nonconsumptive human activity (e.g., recreational hiking and camping) can seriously affect natural ecosystems.

Recreational activities are the principal reason for human intrusion into natural environments. Hiking and camping have a minor but significant impact on natural forests, rangelands, and desert ecosystems. The amount of disturbance is proportional to the volume of activity and the proximity to population centers; access by roads is the determining factor. Cole (1989) has estimated vegetation loss as a result of camping, concluding that his sample campsites had absolute vegetation losses of 37 to 85%. Off-road vehicles (ORVs) can have even more severe impacts on local terrestrial habitats. In particular, ORV races can devastate fragile desert ecosystems. These environments are very slow to recover and often include rare endemic species. In addition to many rare plant species, the endangered desert tortoise is at risk. Skiing and other winter sports are examples of activities that impact relatively isolated mountain areas. These activities are often accompanied by the more deleterious effects of land use conversion into resort development.

Military maneuvers and other training or testing activities can also have significant impacts on terrestrial environments. Bird communities and certain small mammal populations were negatively affected by Army training maneuvers in the Mojave desert (Krzysik 1984). The management of military installations in the Southeast has serious implications on the survival of the endangered red-cockaded woodpecker. Both physical disturbance (especially from tracked vehicle activity) and noise contribute to habitat degradation from military activities.

Exotic species have been introduced into natural areas for game hunting, and as biological controls for other pest species. Accidental releases have also had major negative impacts on natural habitats and native species. Indeed, the entire eastern deciduous forest ecosystem has been permanently altered by the chestnut blight; the loss of tree mast likely precipitated the extinction of the common passenger pigeon. Similarly, the outbreak of dutch elm disease also contributed to the degradation of riparian habitats in the Midwest. Today, severe habitat impacts from exotic species are most prevalent in Hawaii.

The Hawaiian archipelago has lost more than 75% of its original endemic land bird fauna through prehistoric and historic extinctions; the comparable Galapagos archipelago as a whole is not known to have lost a single land bird species (Loope et al. 1988). The aboriginal Hawaiians converted most of the land below the 600-meter elevation to agriculture on the eight main islands. Subsequently introduced species and factors contributing to habitat destruction include herbivorous mammals (goats and pigs), predation by ants, frequent and intense fires, dogs, cats and mongoose, alien arthropods, mollusks, and alien plants. More than 80 vascular plant species in Hawaii currently pose threats to the native biota.

Types of Impact to Habitats

The degrees of impact caused by each of the aforementioned activities varies both within and among different kinds of activity. The level of impact is determined both by the intensity and extent of the activity, and by the specific type of impact on the habitat of concern. The impacts to habitats, and to their values and functions, from the activities discussed in the previous section fall into four general categories:

- Destruction of habitat.
- Fragmentation of habitat.
- Simplification of habitat.
- Degradation of habitat.

The nature of these impacts depends on the specific stress created by each activity. In most cases, a single activity will include several **stressor processes** that impact habitat. For example, the activity of logging a forest includes removal of the trees, associated drying of the forest floor, erosion and sedimentation of nearby streams, and disturbance from noise and human activity. The major stressor processes affecting habitats include the following:

- Vegetation removal.
- Dehydration and inundation.
- Erosion, sedimentation, and soil compaction.
- Eutrophication.
- Acidification.
- Salinization.
- Thermal warming.
- UV-B exposure.
- Contaminant toxicity.
- Noise and visual disturbance.
- Introduced species.

These stressor processes can result in the following effects on habitat:

- Direct mortality of resident species.
- Physiological stress and decreased reproduction.
- Disruption of normal behavior and activities.
- Segmentation of interbreeding populations.
- Modified species interactions and alien species invasion.

Although all of the stressors affecting habitat can have serious impacts, physical alteration of habitat has eclipsed intentional and incidental taking as the major cause of population reduction among species. At greatest risk are the following groups of species: large terrestrial mammals, bats, hole- and ground-nesting birds, amphibians, snails, conifers, herbs, grasslands, freshwater stream organisms, river fishes and mollusks, and estuarine vegetation (Norse 1990b).

Traditional impact analyses have concentrated on degradation of habitats from contamination. The focus of this analysis is on the loss and degradation of habitat through direct conversion and exploitation of the ecological resources. Although these stressors usually have a much greater impact, additional impacts from contamination should also be considered. In addition, it is important to consider the cumulative impact of multiple effects and the indirect effects of activities. The following sections discuss the different kinds of impacts on habitat.

Destruction

The ultimate form of habitat degradation is the destruction of a natural ecosystem through its "conversion" to another land use. In each conversion, the original natural characteristics of the land are eliminated, while the associated habitat values are modified to varying degrees. Occasionally, wildlands (providing ecosystem services and wildlife values) that have been converted to managed lands (providing harvestable timber or agricultural crops) can be restored to a similar, although not identical, natural state. In contrast, lands converted to urban or industrial uses virtually never recover their ecosystem integrity or habitat values.

Physical alterations of many kinds cause habitat destruction. In terrestrial environments, the clearing of vegetation (trees, shrubs, grasses) is the principal stressor. The greatest impacts occur when vegetation removal is accompanied by leveling operations (that destroy the original topography and soil profile) and building or road construction (covering the area with permanent structures). The burning of vegetation and the creation of landfills for waste disposal are other means of destroying terrestrial habitats. Clear-cut logging and severe overgrazing can also clear habitats of native vegetation.

In wetland environments, filling and draining operations destroy wetland habitats and create modified terrestrial habitat, while impoundments flood wetlands to create deepwater aquatic systems. As with terrestrial environments, the construction of buildings or roads can eliminate wetlands. The extraction of peat can also destroy wetlands. In aquatic environments, the inundation or diversion of water through flow alteration (via damming or channelization) is the principal means of eliminating habitat. Dredging, filling, and draining also destroy aquatic habitat.

Fragmentation

While all the activities mentioned in the previous section can result in the destruction of entire habitat types, they often only destroy part of a habitat, leaving other areas intact. Depending on the scale of concern, many instances of local habitat destruction are better thought of as habitat fragmentation. The interruption of a river with a reservoir, the clearcut logging of mature forest, and the building of a road through a salt marsh are all examples of habitat fragmentation (Norse 1990b).

Such fragmentation is the principal cause of loss of "area-sensitive" species (Harris 1985) and the most serious threat to biological diversity (Wilcox and Murphy 1985; Harris 1988). The consequences of habitat fragmentation (Harris and Atkins 1990) include the following:

- Amplification of mortality and inbreeding (i.e., risk to sedentary species from random variation in demographic and genetic variables when isolated).

- Extinction of wide-ranging species (e.g., wolves, black bears, panthers, manatees).

- Loss of interior or area-sensitive species (e.g., sharp-shinned hawk, Cooper's hawk, Swainson's warbler, red-cockaded woodpecker).

- Erosion of genetic diversity from within rare species.

- Increased abundance of weedy species (regionally distinct communities give way to globally homogeneous ones).

As an example, only 2 of 11 native large mammals in Florida (the raccoon and white-tailed deer) are doing well in the face of increasing fragmentation of natural habitats. Other examples of negative impacts from fragmentation include the spotted owl; the Spotted Owl Committee proposed that habitat conservation areas (HCAs) be linked by forests with a minimum canopy closure. Studies in Maryland, Michigan, and Oregon show that the occurrence of most forest-dependent species is correlated with forest size; contiguous forests of 100 to 300 ac are needed by area-sensitive birds, primarily long-distance, insectivorous, neotropical migrants, such as flycatchers, vireos, and wood warblers (Jahn 1991).

Simplification

Habitat simplification includes the removal of ecosystem components such as standing dead trees, cover logs, or stream debris; the death of sensitive submerged plants from siltation; and the loss of microhabitats (such as nests and dens) that are rendered unusable by human intrusion. Universally, the removal of vertical habitat structure reduces the diversity of species. Structural diversity provides more microhabitats (e.g., nest sites) and allows for more complex species interactions (e.g., avoidance of predation and partitioning of foraging space).

While forest clearcutting is both a form of destruction (for the forest stand) and of fragmentation (for the forest watershed), selective logging of preferred tree species is a form of habitat simplification. This is in contrast to timber harvesting practices that are nonselective and often closely mimic natural stand conditions. During selective cutting, not only does the composition of tree species change, but logging creates more extreme microclimates that are usually hotter, colder, drier, and windier than in natural forests. The immediate impact on resident species is the desiccation of forest plants, fungi, slugs, and salamanders that require moist conditions (Norse 1990b).

Within rangeland systems, ecosystem integrity is maintained through the balance of native grass and shrub species. Grazing by domestic livestock can selectively remove species and facilitate the invasion of exotics. In most cases, the proliferation of nonnative species results in habitat simplification that is detrimental to native birds and other wildlife.

Degradation

Degradation of habitats can include the fragmentation or simplification of habitat structure, but more specifically refers to a decrease in the health or ecological integrity of the "intact" habitat. Chemical contamination resulting from air or water pollution is a significant cause of habitat degradation. Although toxic effects may be the most severe, conventional pollutants and other effects may exist in greater frequency and extent. For example, soils are degraded through erosion or soil compaction. Lakes are particularly sensitive to eutrophication and acidification. Rivers and streams can be degraded by nutrient enrichment, as well as siltation and turbidity. Salinization and salt water intrusion also degrade habitats, as do temperature modification and noise. Underground water sources and their contributions to ecosystem integrity can be degraded by activities, such as irrigation and mineral mining, that result in the draw down of aquifers. The invasion of exotic plants and animals can seriously degrade natural systems through modified species interactions. Global climate change, including increased temperatures and UV-B exposure, has the potential to degrade habitats of all kinds.

Vulnerability to Impacts

The impacts of degrading activities on habitat depend on the vulnerability of the habitat and the relative contributions of other cumulative and interactive impacts. A habitat's sensitivity is determined by its resistance to change (i.e., its ability to resist degradation) and its resilience (i.e., its ability to recover its original condition) (Westman 1978). Resistant habitats often have intrinsically stable and fertile soils, moderate rates of water movement, mild climate regimes, and food webs that are functionally diverse and contain individuals or species preadapted to the particular stress. Resilient habitats are often topographically low and proximate to unstressed habitats containing highly mobile colonizers (Sedell et al. 1990).

Species are usually more vulnerable to anthropogenic impacts if they possess small effective population size, narrow geographic distributions, large area requirements, specialization, intolerance of disturbance, large size, slow reproductive rate, evolutionary naivete, or "amphibious" habits (Norse 1990b). Vulnerability characteristics of habitats or ecosystems (and the stressor to which they are vulnerable) are listed below:

- Impermanence (suppression of fire frequency).
- Oligotrophy (alteration of nutrient cycling).
- Undersaturation (biological invasion).
- Isolation (elimination of recolonization).
- Small size (impacts on edges).
- Proximity to human populations (disturbance).

The undersaturated naive biotas of the Hawaiian Islands and southern Florida are especially susceptible to many stressors, including invasion by exotic species. All habitat areas are vulnerable to unprecedented permanent major changes in environmental conditions. Unlike periodic natural disturbance (such as fires, windthrow, and flooding), global atmospheric change (e.g., warming and increased UV-B or CO_2) and the introduction of alien species pose challenges beyond the capabilities of most natural systems. Perhaps the greatest threat to biodiversity is the impending interaction between climate change and habitat fragmentation.

General Mitigation Procedures

Appropriate measures for the mitigation of habitat loss or degradation depend on both the habitat type and the specific degrading activities, stressor processes, and habitat impacts. Specific mitigation information is provided in the regional sections of this document. In this section, general considerations for habitat mitigation are discussed. For a mitigation to be successful, the ecological integrity of the habitat must be maintained. This can be accomplished directly by preservation measures that avoid impacts. In other cases, careful mitigation plans can reduce or eliminate impacts on the integrity of the habitat.

Habitat Integrity

Traditionally, mitigations have concentrated on species-specific habitat components such as the availability and appropriate interspersion of cover, food, and water. Other species requirements include protein-rich foods, den or nest sites, and territorial spacing or colonial clustering, and may vary seasonally, especially among migratory waterfowl and anadromous fish. Greatest attention has been paid to the diversity of habitat structure, both vertical layering and horizontal edge or transition zones, that provide for greater species and ecological diversity. While these considerations are appropriate for mitigations focusing on certain species or individual site diversity, they do not incorporate landscape-level concerns for regional diversity. For the purposes of this document, mitigations of habitat degradation will focus on the ecological integrity of the habitat of concern and not on the species or diversity components that may be desirable from a wildlife management point of view.

Recent research has indicated that floristic (plant species) diversity is superior to structural (number of vegetation layers and patches) diversity as an indicator of wildlife distribution. This emphasizes the need to avoid oversimplification in habitat analyses and to look at the detailed ecology of each habitat and define it in precise ecological terms. Natural habitats are dynamic ecological systems that require natural patterns of disturbances. Proper mitigation plans must provide for natural habitat heterogeneity in time and space. An important tool for providing natural disturbance patterns is fire management. Proper use of controlled fires can be an effective mitigation of the impact of fire suppression in managed areas.

Mitigation for habitat conservation must ensure that the cumulative impacts of all activities within the landscape (perhaps over areas of 10,000 to several 100,000 ac) are addressed to maintain ecosystem integrity and health. The preservation of individual habitat areas is often not sufficient to maintain the ecological integrity of the greater ecosystem. In addition, the size, diversity, and distribution of key habitat tracts must be conserved to provide for the natural diversity characteristic of the larger eco-complex or region. Finally, unique ecosystems (such as islands) may require unique mitigation solutions (Samson et al. 1991).

<u>Mitigation Guidance</u>

Mitigations to address the habitat impacts of destruction, fragmentation, simplification, and degradation include the following four measures (modified from Flather and Hoekstra 1989):

1. Preservation
   - Outright purchase or set aside of land
   - Partial purchase through conservation easements, long-term leases, or management agreements.

2. Management practices
   - Rotation and method of timber harvesting
   - Timing and extent of grazing
   - Control of pollution
   - Elimination of structures.

3. Restoration
   - Direct manipulation through seedings, plantings, physical or chemical treatment
   - Creation of wetlands
   - Control of pollution
   - Removal of barriers to fish migration
   - Control of livestock access to riparian areas.

4. Compensation
   - Purchase of lands of comparable habitat size and quality
   - Provision of financial restitution.

A more detailed set of Mitigation Means and Measures (in general priority order) has been devised by the Fish and Wildlife Service for mitigation development related to fish and wildlife and their habitats (FR 46(15):7660, 1981). This list is provided in the accompanying box.

U.S. Fish and Wildlife Service
Mitigation Means and Measures

A. Avoid impact
1. design project to avoid damage or loss including management practices such as
   - timing of activities or
   - structural features such as multiple outlets, passage or avoidance structures, and water pollution control facilities
2. use nonstructural alternative
3. cancel project

B. Minimize impact
1. include conservation of fish and wildlife as authorized purpose
2. locate a least environmentally damaging site
3. reduce the size of the project
4. schedule timing and control of construction and maintenance to minimize disruption of biological community structure and function
5. use selective tree clearing or other habitat manipulation
6. control water pollution through BMPs
7. time and control flow diversions and releases
8. maintain public access
9. control public access for recreational or commercial purposes
10. control domestic livestock use.

C. Rectify the impact
1. regrade disturbed areas to contours for optimal habitat or original condition
2. seed, fertilize, and treat areas to restore fish and wildlife
3. plant shrubs and trees and other vegetation to speed recovery
4. control pollution spoil areas
5. restock fish and wildlife in repaired areas

D. Reduce or eliminate the impact over time
1. periodically monitor to ensure continuous operation of mitigation
2. train personnel properly to preserve fish and wildlife
3. maintain or replace equipment or structures to prevent loss due to equipment/structure failure

E. Compensate for impacts
1. conduct wildlife management activities to increase habitat values; give priority to project lands and nearby public lands
2. construct habitat to fully restore and rehabilitate altered habitat, or to modify existing habitat suited "evaluation species" to completely offset habitat value losses
3. designate fish propagation facilities
4. designate legislative set-aside or protective designation for public lands
5. designate buffer zones
6. lease habitat
7. acquire wildlife easements
8. acquire water rights
9. acquire land in fee title

Mitigation Principles

The development of specific mitigation plans must be based on a thorough understanding of the site conditions and the activities impacting habitats. Nonetheless, certain basic principles of ecological management should be followed when specific mitigation measures are developed. The following seven general mitigation principles apply to all habitat conservation efforts:

1. Base mitigation goals and objectives on a landscape-scale analysis that considers the needs of the region.

2. Mimic natural processes and promote native species.

3. Protect rare and ecologically important species and communities.

4. Minimize fragmentation of habitat and promote connectivity of natural areas.

5. Maintain structural diversity of habitats and, where appropriate, species diversity to promote the natural variety of the area.

6. Tailor management to site-specific environmental conditions and to the unique impacts of the specific degrading activity.

7. Monitor for habitat impacts from activities and revise mitigation plans as necessary.

A landscape or ecosystem-level perspective is central to these principles. R. Max Peterson (Emeritus Chief of the Forest Service and Executive Vice President of the International Association of Fish and Wildlife Agencies) stated that "when land is cleared, care must be taken to maintain the minimum size areas of sensitive habitats, with buffers and corridors as needed to ensure the integrity of the landscape ecosystem" (Giltmier 1991). The concept of providing for landscape integrity when habitats are fragmented is central to habitat mitigation in forest, rangeland, wetland, and aquatic systems. The two most important methods for maintaining the integrity of fragmented habitats are (1) the provision of buffer areas, and (2) the creation of habitat corridors. Buffers represent the principal method of avoiding impacts to sensitive areas, and habitat corridors provide the best means of mitigating habitat isolation. The most common means of creating both buffer areas and corridors is the preservation of natural habitat along streams, steep slopes, and other sensitive areas.

Habitat Buffers

The preservation of a habitat of concern includes both the avoidance of direct conversion of the area and the maintenance of adequate buffer areas so that edge effects and other negative impacts do not affect the sites. For example, powerline corridors through forests can be "feathered" to avoid some edge effects (Gates 1991). Additional areas adjacent to the corridor can be cut to create successional bands of vegetation parallel to the corridor opening; this reduces predation rates at the edge and minimizes the barrier effects. However, a wider edge results in less forest interior.

Mitigation procedures for many projects can be designed to reduce the effective width of a cleared area and thus decrease the barrier effect. These include creation of small lobes or peninsulas of dense vegetation reaching into the open area, or the creation of entire breaches across the area, either by leaving the habitat intact or by staggered defoliation regimes. The establishment of a stable shrub community in a forest corridor can provide movement by less mobile animals with small home ranges (Niering and Goodwin 1974).

Research into the impacts on benthic invertebrate communities of streams indicates that buffer strips of at least 30 m are required to prevent alteration in invertebrate diversity and ecological structure (principally the increase in abundance of pollution-tolerant taxa such as chironomids). These buffer strips serve to maintain riparian canopy and stream channel stabilization. Failed road crossings also negatively impact stream ecosystems (Erman et al. 1977).

Habitat Corridors

Mitigation of habitat fragmentation involves the restoration of habitat "connectivity" (Norse 1990b). To address the effects of fragmentation, conservation biologists are calling for increased provision of habitat corridors. Unlike untested management plans based on island biogeography theory, corridors have been used successfully in wildlife management for 50 years (Harris and Atkins 1990). Corridors provide for the movement of animals, serve as a population source, contain whole communities, and withstand natural disturbance events, but they also provide for contamination transmission (Csuti 1991). Because edge effects reach 200 to 600 m into the forest, Pace (1990) recommends a minimum corridor width of 6.4 km to mitigate edge effects.

In a landmark court decision concerning the USDA Forest Service timber sales in the Klamath National Forest, federal agencies were required to consider an area's importance as a "biological corridor" linking wilderness areas before permitting logging. The resultant Klamath Corridors Proposal can serve as a model of habitat fragmentation mitigation (Pace 1990). It recommends connectivity as superior to isolation, continuity over fragmentation, and creation of larger rather than smaller corridors.

Mitigation Measures

The first priority in developing mitigation plans for habitat loss or degradation should be avoidance of the impact. This is usually a siting issue, where construction operations and degrading activities are located at a distance from the habitats of concern. The habitat is adequately preserved if all possible impact scenarios are accounted for. Barring this solution, effective management measures must be implemented to ensure the protection of the habitats of concern. Failing effective management, mitigation falls to the restoration of habitat, which is often problematic, or finally to compensation.

Restoration activities will not be discussed in this document, although they are receiving increased attention as mitigation measures, especially in wetland and aquatic systems. The recent volume produced by the National Research Council (1992) provides a comprehensive discussion of the science, technology, and public policy involved. Many of the principles espoused in this book also apply to terrestrial systems.

This document focuses on the general management practices that can be undertaken to mitigate habitat degradation and loss resulting from activities in forest and rangeland environments. A central tenet of the management approach to habitat mitigation is the control of pollution. This is especially true for wetland and aquatic systems where, after physical alteration, off-site impacts to hydrology and water quality pose the greatest threat. There is also a growing body of literature on best management practices

(BMPs) as mitigation measures for aquatic systems. Notably the nonpoint source, clean lakes, and national estuary programs of EPA are promoting BMPs to protect sensitive habitats. Many of these measures apply to wetlands and are being implemented under section 404 of the Clean Water Act and provisions of the Coastal Zone Management Act. The reader should refer to these programs for additional information on mitigating impacts to wetland and aquatic systems.

In contrast to aquatic systems, forests and rangelands are primarily threatened by direct exploitation of their resources (trees and forage grasses). Specific guidance on mitigation measures is provided in each regional habitat evaluation section. The following discussion addresses general mitigation issues for timber harvesting and grazing methods.

Timber Harvesting Mitigation Methods

At a minimum, the production of commercial wood products from an area must not exceed the sustainable level if the ecological integrity of a forested area is to be maintained. Where sensitive forest types exist, logging may be completely prohibited or constrained to specific methods to prevent habitat loss or degradation. In other areas, more extreme harvesting methods may be allowed or prescribed to establish or maintain desired forest conditions. Acceptable methods will vary according to local forest ecology and the desired future condition of the site. Analysis of harvesting techniques must be based upon an analysis of the structure and diversity of the forest canopy, midstory, and understory.

A recent directive of the Chief of the U.S. Forest Service acknowledges this fact and points out that clear cutting is acceptable only when needed to replicate natural ecological processes. Selective cutting can preserve forest structural diversity, the primary determinant of wildlife habitat (Harris et al. 1979). However, it can reduce horizontal diversity (NRC 1982). The harvesting technique employed must be based upon sound silvicultural prescriptions and demonstrate its capability to maintain vertical diversity (foliage height diversity), horizontal diversity (interspersion, edge, juxtaposition, patchiness), and a mixture of live and dead wood. Specific timber harvesting operations should be designed to preserve the structure and diversity of the natural forest habitat.

Grazing Mitigation Methods

The current degraded state of rangelands requires restoration as well as management plans. In both cases, the timing and extent of continued grazing will determine whether range conditions worsen or improve. Increased irrigation for agriculture may delay improvements by adversely affecting water tables and stream flow on rangelands. Rest-rotation grazing can improve range conditions, while intensified chemical use and mechanical brush removal will likely further degrade range habitats. The future management of riparian areas will have the greatest impact on rangeland wildlife and ecosystem health (NRC 1982).

In the past, range condition has been estimated by forage production or production of livestock. More recently, condition has been based on the deviation from an ideal range condition or ecological climax. More effective use of ecological analyses of range condition will improve the management of rangelands. In particular, range managers need the following tools (Wald and Alberswerth 1989):

- More data (range condition is unknown on many rangelands).

- Management plans for each site (these should be ecologically based and site specific).
- More management resources.
- Commitment from management to implement grazing reductions or riparian habitat improvement.

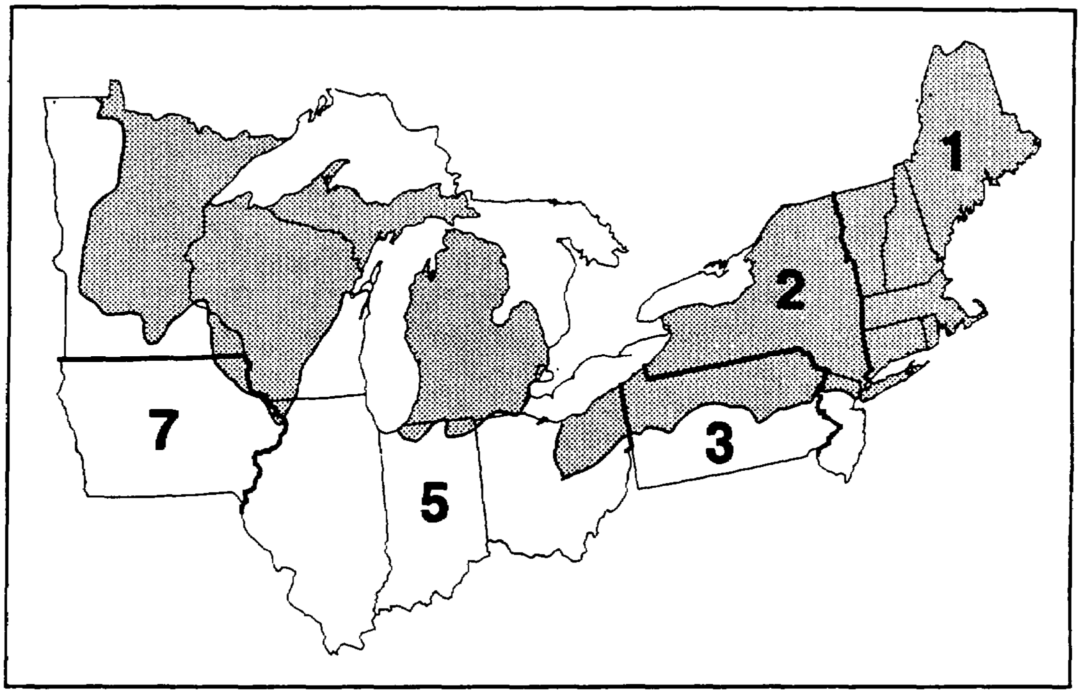
Monitoring for Mitigation Compliance

Successful mitigation of habitat impacts requires that the proposed mitigation measures are effectively implemented and maintained. However, the consideration of habitat effects is often hampered by information gaps and limits to predictive capability. Therefore, it is essential that all mitigation plans include adequate provisions for baseline and post-project monitoring of habitat conditions.

The fact that many restoration projects designated as mitigation have not achieved their desired objectives is well documented. It is also believed that mitigation measures for many projects are not adequately implemented or enforced. Therefore, determination of the true effectiveness of mitigation should be the goal of monitoring programs. The following ten-step process for monitoring mitigations for habitat impacts has been modified from Noss (1990):

- Establish objectives of the mitigation.
- Gather and integrate data.
- Establish baseline conditions.
- Identify elements at risk.
- Formulate specific questions to be addressed by monitoring.
- Select indicators.
- Identify control areas and treatments.
- Design and implement the sampling scheme.
- Validate relationships between indicators and endpoints.
- Analyze trends and recommended management actions.

North Habitat Region: *Northern Lakes and Forests*



Geographical Description of the Region

The North Habitat Region, *Northern Lakes and Forests*, contains all of eight states and parts of eight additional states. The region includes all of Maine, New Hampshire, Vermont, Massachusetts, Connecticut, Rhode Island, New York, Michigan, and parts of New Jersey, Pennsylvania, Ohio, Iowa, Indiana, Illinois, Wisconsin, and Minnesota. EPA Region 1 is included in its entirety; parts of EPA Regions 2, 3, 5, and 7 are also included. The accompanying map indicates the boundaries of this habitat region and the states it comprises.

The *Northern Lakes and Forests* comprises eight ecoregions (Omernik 1987). The vegetation of this region includes northern hardwoods (maple, birch, beech, hemlock), elm, ash, Great Lakes spruce and fir, Great Lakes pine, conifer bogs (spruce, larch, arborvitae), maple, basswood, and oak savanna (oak and bluestem). The land use patterns include swamps, marshlands, forests and woodlands (mostly ungrazed), croplands, croplands with pastures, and urban.

Habitats of Concern

The *Northern Lakes and Forests* contains many habitats of concern, of which the most obvious fall into the three general categories of old-growth forest, barrens, and Great Lakes ecosystems. The principal habitats of concern most at risk in the *Northern Lakes and Forests* are listed below.

PRINCIPAL HABITATS OF CONCERN
IN THE *NORTHERN LAKES AND FORESTS*

1. Old-growth and mature forests
   - northeast conifer and hardwoods forests
   - central hardwoods forests
   - boreal forests of northern lake states

2. Barrens
   - pitch pine-scrub oak barrens
   - Appalachian shale barrens
   - other cliff and ridge talus, tundra, meadow, and heath communities

3. Great Lakes coastal habitats
   - barrier islands
   - dune systems
   - coastal wetlands
   - pannes or intradunal ponds
   - rocky shores along Lake Superior with arctic species
   - bluffs with oak savannas, jack pine woodlands, and beech-maple forests

Habitat Values and Trends

The *Northern Lakes and Forests* originally consisted of a vast forested area covering both New England and the northern Lake States. Once virgin forest, New England was cultivated on 75% of arable land by 1840, but is now primarily forested again (DeGraaf 1991). The White Mountains of New Hampshire and western Maine contain many forest cover types; northern hardwoods constitute approximately half of the area. Because of the glacial origin of soils in New England, many of the most fertile sites are on midslope and produce hardwood forests. The impervious layer (fragipan) underlying much of these till soils produces vernal pools, seeps, and wet ground during the spring even on upper slopes. Therefore, the forest landscape of New England is a mosaic of forest types and nonforest habitats that occur in relatively small patches, especially in the mountains. Among these isolated habitats are various forms of barrens that support numerous rare species. The vegetation of the northern lakes region has a more recent history of timber harvesting and forest regrowth, but consists of a greater variety of habitats including many northern forest types, coastal habitats, and wetland types.

Northeastern Forests

Forests of seven northeastern states comprise 49.5 million ac or about 70% of the total regional land area (Barrett 1980). Major forest type groups are maple-birch-beech, white-red jack pine, spruce-fir, loblolly-shortleaf pine, and oak-hickory. By 1890, most of the northern spruce had been cut; the hardwood forests soon followed. Large fires swept over northern New England shortly after the turn of the century. Other forest losses have been due to the chestnut blight, diseases of birch and beech, and gypsy moth attacks on oak. Overcutting of commercially desirable species has resulted in the expansion of elm-ash-red maple at the expense of beech-birch-sugar maple. In the Northeast, substantial areal declines have occurred in oak-gum-cypress (53%), loblolly-shortleaf pine (49%), elm-ash-cottonwood (38%), aspen-birch (25%), oak-hickory (20%), and spruce-fir (14%) (Flather and Hoekstra 1989).

In the last 100 years, one-fifth of the region's total acreage has reverted from pasture and tillage to brush and forest (Hagenstein 1990). Since the 1950s, most of the increase in forest area is directly linked to the decrease in farm area, especially dairy farming. By the 1960s, the areal extent of suburban developed land surpassed that of agricultural land. Since that time, the development of recreational homes in the mountains and along coasts and lakeshores has resulted in large areas of fragmented, sensitive lands. This process has fragmented ownership in a region with the lowest ratio of publicly owned land of any forested region in the United States. The result of this long history of exploitation is that less than 1% of New England's total acreage is in pristine ecosystems (Giltmier 1991).

Extended wildfire protection and insect and disease control programs have greatly reduced the loss of forest trees to these factors. However, both mortality and lowered growth rates have resulted from air pollution in the Appalachians and eastern Canada. Projections indicate a decline throughout the North over the next 50 years. Urban area has doubled, and small forest parcels and low-value timber lands will likely be converted to other uses. However, several states in the North have adopted regulations to ensure the regeneration of logged areas and to protect water quality (Hagenstein 1990).

Northern Lakes Forests

Approximately 43% (52 million ac) of the total area of Michigan, Minnesota, and Wisconsin is forested (Barrett 1980). Replacement of forest with agriculture increases from the East to the West and from the North to the South. The Lake States forests are 75% hardwoods (principally aspen-birch) and 25% conifer (mostly spruce-fir). They include 15 northern forest cover types, 4 central hardwood cover types in the "big woods" area of Minnesota and the southern portions of Wisconsin and Michigan, and 8 boreal forest cover types.

In 1902, the region led the country in timber production; by 1910, the majority of commercially valuable white and red pine was gone. In later years, overexpansion of farming cleared vast areas of forest. Fires and swamp drainage also contributed to devastation of the forest area in the region. Substantial losses are still occurring in the forested areas in the northern Lake States. Logging is proceeding at a rapid rate in Michigan. In Wisconsin, oak forests are being intensively harvested for oak veneer, and aspens are declining as a result of forestry management practices. White pine and hemlock in southern Michigan, once dominant in the area, are today nearly absent. The elm-ash forest type in

Ohio, Indiana, and Michigan has been reduced by 90% as a result of conversion to agriculture and urbanization.

Forest Values

Forest ecosystems support 90% of the total bird, amphibian, and fish species and 80% of mammal and reptile species in the United States. In addition, the *Northern Lakes and Forests* contains an average 2.6 endangered and threatened species per county as of 1984 (Flather and Hoekstra 1989). Some of the ecological values of each of the regional forest types are listed below:

- Oak-hickory - supports southern bald eagle, red wolf, red-cockaded woodpecker and contains many diverse mesic environments.
- Maple-beech-birch - includes a wide variety of tree, shrub, and forb species that provide aesthetic, wildlife (e.g., moose), and recreational resources.
- Spruce-fir - contains many remote and pristine environments that support moose, great horned owl.
- Aspen-birch - represents a pioneer community that follows disturbance and supports ruffed grouse and moose.
- White-red-jack pine - supports threatened species such as eastern timber wolf, peregrine falcon, and Kirtland's warbler.
- Elm-ash-cottonwood - represents important riparian habitat along moist river and stream bottoms, and in and around swamps and depressions.

Old-Growth and Mature Forests

Old-growth forests are unique, vanishing environments that merit preservation for aesthetic, ecological, and scientific values (Society of American Foresters 1984). Although the *Northern Lakes and Forests* do not contain the acreages of virgin forest still found in other parts of the country, many mature forests greater than 100 years old do exist. These mature forests possess a variety of important ecosystem values and should be the focus of habitat conservation efforts.

As an example, the majority of remaining old-growth stands in Pennsylvania are on steep mountain slopes and deep, narrow, boulder-strewn ravines. This is a result of a long history of natural disturbance and anthropogenic degradation that has dramatically changed the composition of the present day oak forests of Pennsylvania. They differ dramatically from the original types that were present before settlement in early 1600s. Even with extensive clearing for agriculture and coal mining, the state was 75% forested in early 1800s. By 1850, however, Pennsylvania was the logging center of nation. Subsequent attacks by the American chestnut blight and beech bark fungus and severe vegetation destruction from growing white-tailed deer populations killed many trees. Most important, extensive clearcutting caused a shift in species composition with declines in white pine and eastern hemlock and increases in yellow birch, black cherry, and red maple. Remaining old growth in Pennsylvania can be classified into four types after Kuchler (1964): beech-maple; hemlock-northern hardwood forest (hemlock-white pine-beech-black birch); Appalachian oak forest (chestnut oak-white oak-red oak-hickory); and mixed mesophytic forest (white oak-red oak-yellow poplar-basswood) (Smith 1989).

The old-growth forests of the northern Lake States are another important habitat type. Historically, pine and hemlock-northern hardwood forests were most extensive. Nontraditional old-growth ecosystems include northern white-cedar, speckled alder, northern pin oak, black ash, bigtooth aspen, and trembling aspen. The old-growth forest ecosystems of this region contain a greater regional and local diversity than has been generally appreciated (Barnes 1989). *Northern Lakes and Forests* habitats vary with the pattern of structurally (physiography, soil, vegetation) and functionally different landscape ecosystems.

Great Lakes Coastal Ecosystems

The many wetland and sand dune ecosystems of the Great Lakes coastal region are important in the *Northern Lakes and Forests* and vary according to physiography, associated soils, and other abiotic factors (Barnes 1989). Many of the ecosystems that have not been destroyed or highly modified are imminently threatened. Impacts include the lumbering of most Great Lakes forests in the late 1800s, destruction of over half of the wetlands, pollution from heavy industry, and the proliferation of lakefront residences and structural modifications to protect shoreline property. Degradation from recreational use and the accidental or purposeful introduction of alien species are also important (Hiebert 1990).

Wetlands and Aquatic Systems

Because the Great Lakes contains 54% of the nation's water area (a total of 58 million ac), wetlands and aquatic systems are especially important habitats in the *Northern Lakes and Forests*. Along the Great Lakes, large inland coastal marshes lie behind beach ridges and are often influenced by lake water levels and wind tides. Other wetlands are eutrophic or boglike and, although still common, are much reduced in size (e.g., 71% of Michigan marshes have decreased in area). In addition, many glacial wetlands occur within the northern forests. They are often surrounded and invaded by trees producing boglike edges with sedges and mosses and alder willow.

Many smaller lakes exist throughout the *Northern Lakes and Forests*. Acidification from atmospheric deposition has had a severe impact on lakes of the Northeast. Diverse marine environments exist along the northern Atlantic coast, including many glaciated estuaries and the modified Hudson River Valley.

Activities and Impacts Affecting Habitats

The major sources impacting habitats in the *Northern Lakes and Forests* include residential developments, industrial and commercial developments, dam construction, interstate highway or expressway construction, logging and silvicultural practices, solid waste disposal, and peat mining. These activities have had adverse impacts on species populations and their behavior, as well as on ecosystem processes such as energy flow and nutrient cycling. They have also contributed to the proliferation of nuisance plants and animals. In its comparative risk analysis, EPA Region 1 concluded that the highest risk to upland and aquatic habitats in New England is concentrated in rapidly growing areas (e.g., central Connecticut, southern Maine, and New Hampshire). Historical losses of terrestrial environments are greater toward the coast and the southern part of the region.

Agricultural conversion and grazing are relatively minor activities in the region, while timber harvesting practices and peat mining continue to degrade terrestrial environments throughout the region, especially in Maine and the northern Lake States. However, the conversion of land to industrial, residential (including second homes), and transportation uses is the most severe cause of terrestrial habitat loss in the *Northern Lakes and Forests*. The major metropolitan areas in this region are under enormous pressure from human populations, and the effects are degrading the remaining natural habitats in the area.

The following activities result in the major impacts on habitats of concern in the *Northern Lakes and Forests*.

IMPACTS ON HABITATS OF CONCERN IN THE *NORTHERN LAKES AND FORESTS*					
	Land Conversion	Timber Harvesting	Grazing	Water Management	Other
Mature forest	Destruction from residential development	Simplification by replacement with plantation species	Minor	Minor	Degradation with increased recreation pressure
Barrens	Destruction from residential development	Degradation	Degradation	Minor	Degradation with increased recreation pressure
Great Lakes systems	Destruction from residential development	Moderate	Minor	Destruction of wetlands	Degradation with increased recreation pressure

Land Conversion

Historically, land conversion of both uplands and wetlands has profoundly affected the natural communities in the Northeast. The early clearing of eastern forest for small farms benefitted robins, woodchucks, and bobwhite quail, but negatively impacted wild turkeys, black bears, and moose. Since that time, the large population centers are primarily responsible for the conversion of natural areas, i.e., through industrial and residential development. Because cities concentrate on coastal areas, the unique environments of the Atlantic and Great Lakes shores have been most affected. Recent increases in second home and resort development are now contributing to construction in previously pristine areas. In many cases, rare barrens, dunes, and wetlands areas are being converted with the loss of many rare plant species. More generally, "spin-off development" associated with highway construction has facilitated the expansion of land use conversions into rural areas. This increased road construction is causing severe fragmentation of terrestrial habitats.

Timber Harvesting

Timber harvesting activities can fragment, simplify, and degrade forest habitats. The faunal communities inhabiting forests vary with the successional, or seral, stage such as grass/forb, shrub/seedling/sapling, medium tree, and large tree. Because the principal impact of timber harvesting practices is to convert forest stands from later to earlier seral stages, logging has a major impact on resident animal as well as plant species. Timber harvesting telescopes plant succession, shortens rotations, compresses seral stages; and decreases the proportion of old growth. For example, old-growth spruce, fir, and white cedar disappear with short rotations in Maine (i.e., reducing wintering grounds for deer). The conversion of hardwoods to conifers creates structurally simplified plantations that reduce structural diversity and wildlife. This has produced a trend away from declining habitat types and toward common habitat types. Management for monotypic even-aged stands causes increases in forest pest damage which often result in large-scale spraying and the accompanying impacts. Timber harvesting activities also impact nearby aquatic systems through erosion and sediment transport.

Second in concern to the decrease in old-growth forests is the general decline in neotropical migrants that breed in eastern hardwood forests. Although the situation is complicated by losses of wintering habitats for long-distance migrants in Latin America, results indicate that species still present in large blocks of forest are absent from small patches (Robbins et al. 1989). Fragmentation of forest habitat from timber harvesting and from land conversions, especially for transportation, appears to be the major cause of these declines (Terborgh 1989).

Recreational Activities

Forest habitats, and especially the many unique barrens, dunes, and wetland habitats in the *Northern Lakes and Forests,* can be negatively impacted by recreational activities. These impacts are usually localized, but can severely affect the hydrology and nutrient cycling regimes of vulnerable habitats. As an example, the annual Canaan Valley motorcross contributes to the degradation of sensitive wetland habitats through soil erosion.

Mitigations of Impacts

The conservation of habitats requires consideration of mitigations for the major activities impacting habitats of concern. In the *Northern Lakes and Forests*, the primary habitat impacts are caused by the following:

- Land conversion and timber harvesting of old growth and mature forests.
- Land conversion of barrens and other rare habitat types.
- Land conversion and pollution of Great Lakes ecosystems.

Land Conversion

Effective mitigation of land use conversion activities can sometimes be obtained only by avoiding impacts on rare or unusual habitat types. Rarely, if ever, is restoration or compensation an adequate mitigation for the loss of these habitats. In these cases, mitigation is a siting issue, where construction and degrading activities are located a distance from the habitats of concern. The habitat is adequately preserved if all possible impact scenarios are accounted for. Barring this solution, effective management measures must be implemented to ensure protection of the habitats of concern.

In the case of barrens habitats or unique Great Lakes ecosystems, hydrological and contamination concerns are especially important. Construction or resource management activities require the use of sediment filter strips and other means of intercepting off-site contaminants. Road building and structural "improvements" must not result in altered hydrological regimes. Where rare plant types exist or where habitats are unstable (e.g., sand dunes), recreational access associated with nearby development may have to be limited.

Amelioration of impacts from land conversion to transportation uses requires special mitigation measures. As with all land conversion, the construction of highways and power-line corridors is primarily a siting issue. Avoidance of sensitive habitats may be accomplished by modifications to the route design, and the extent of disturbance can be limited by careful construction practices. However, fragmentation of the larger area is unavoidable in the case of land conversion to transportation corridors. Structural mitigations can be used to lessen the impact on animal movement across transportation routes. Primarily, these include the construction of fences and underpasses. The goal of these structural measures should be to mimic the natural movement and migration patterns of the affected species.

Timber Harvesting

At a minimum, the production of commercial wood products from an area must not exceed the sustainable level if the ecological integrity of a forested area is to be maintained. Where sensitive forest types exist, logging may be completely prohibited or constrained to specific methods to prevent habitat loss or degradation. In other areas, more extreme harvesting methods may be allowed or prescribed to establish or maintain desired forest conditions. Acceptable methods will vary according to local forest ecology and the desired future condition of the site. Analysis of harvesting techniques must be based upon an analysis of the structure and diversity of the forest canopy, midstory, and understory.

A recent directive of the Chief of the U.S. Forest Service acknowledges this fact and points out that clear cutting is acceptable only when needed to replicate natural ecological processes. Although, selective cutting can preserve forest structural diversity, it can reduce horizontal diversity (NRC 1982). The harvesting technique employed must be based upon sound silvicultural prescriptions and demonstrate its capability to maintain vertical diversity (foliage height diversity), horizontal diversity (interspersion, edge, juxtaposition, patchiness), and a mixture of live and dead wood. Specific timber harvesting operations should be designed to preserve the structure and diversity of the natural forest habitat.

An important component of selective cutting should be the preservation of standing dead trees. Northern hardwood forests contain 24 species of birds that nest, roost, or forage for invertebrates in standing trees with decayed wood. These cull trees are usually the first focus of forest-thinning operations, to the detriment of the birds. Breeding bird abundance declines rapidly following a clear cut, and the species composition continues to change for 10 to 15 years (DeGraaf 1991). However, if trees with cavities are saved, many of these species can successfully forage on sound boles. About one large cavity or den tree per 2 ha is required for populations of large species such as wood ducks; this requires harvest rotations of 100 to 125 years (although rotations of 65 years produce trees large enough for species nesting in smaller cavities).

Responding to the "biodiversity crisis," the U.S. Forest Service is moving toward an ecosystem approach to forest management (Bob Szaro, personal communication). Recent forest management plans have incorporated tenets of the "New Forestry" espoused by Jerry Franklin. These progressive plans require the rigorous implementation of ecological management practices to maintain forest productivity and to preserve the functioning of sensitive forest components such as old-growth or late-successional forests. Effective mitigations for habitat conservation in forest management require specific management measures at the site, watershed, and landscape levels. For example, the location and size of timber harvests should be planned to minimize reduction of the core area of mature forest (e.g., harvest only alternate basins until regrowth). Maintenance of mature-forest stands in managed landscape can be achieved by extending rotation (beyond 80) to 150 to 200 years, by leaving some stands unharvested for old growth, and by linking stands. Landscape-scale considerations include the provision of buffer zones and habitat corridors as discussed in the introduction of this document. The following management measures are recommended for conserving habitat within managed forests:

- *Minimize the construction of new roads and close roads not in use either permanently or* seasonally.

- Use best management practices (BMPs), such as filter strips, to minimize erosion during harvesting or road construction.

- Maintain 100-ft riparian zones with adjacent feathered transition zones to buffer edge effects.

- Restrict harvesting operations to periods when the ground is either dry or frozen.

- Maintain site productivity by retaining large woody material and minimizing mineral soil exposure and compaction during harvesting.

- Manage for natural disturbance patterns to maintain natural openings and successional-stage composition.

- Maintain connections between blocks of interior forest, especially old growth.

- Provide for the protection of special areas, including cliffs, caves, taluses, riparian areas, and old-growth stands.

- Maintain the structural integrity and the native variety of the forest by managing for the natural composition of the following components: vegetative types, seral stages, tree types and sizes, standing dead trees and down material, tree snags, and cavity trees.

Guidelines for Reviewers

Reviewers of environmental impact assessments will find this document useful if they follow the steps laid out in the introduction:

1. Review the status and trends of habitats in the region.

2. Identify the habitats of concern.

3. Link the activities involved with impacts to these habitats of concern.

4. Devise appropriate mitigations for the impacts.

Each reviewer can then determine the adequacy of the environmental impact assessment in question and recommend modifications to enhance its effectiveness.

In identifying the habitats of concern, the reviewer should supplement the information in this document with detailed locational information on the abundance and distribution of habitats within the region of interest, and with any historical information on the extent and quality of these habitats. Most important, the reviewer should characterize the habitats in terms of their ecological values (e.g., use of wooded wetlands by migratory waterfowl).

In considering the links between activities and habitats, the reviewer should look beyond direct impacts to indirect and subtle effects, including cumulative impacts, interactive and synergistic impacts, and scale-dependent impacts (e.g., effects of fragmentation on ecosystem integrity and species home ranges).

In devising possible mitigations, the reviewer should follow the seven principles for habitat mitigation repeated below. The reviewer also should determine whether adequate assurances have been given that the mitigations proposed will be completed.

1. Base mitigation goals and objectives on a landscape-scale analysis that considers the needs of the region.

2. Mimic natural processes and promote native species.

3. Protect rare and ecologically important species and communities.

4. Minimize fragmentation of habitat and promote connectivity of natural areas.

5. Maintain structural diversity of habitats and, where appropriate, species diversity to promote the natural variety of the area.

6. Tailor management to site-specific environmental conditions and to the unique impacts of the specific degrading activity.

7. Monitor for habitat impacts and revise mitigation plans as necessary.

Finally, the reviewer should consider the proposed activities and mitigations in the context of relevant regional program goals and objectives (e.g., whether the outcome of the project will be in accordance with principles set out by regional planning commissions such as those established for the New York Bight and the Great Lakes).

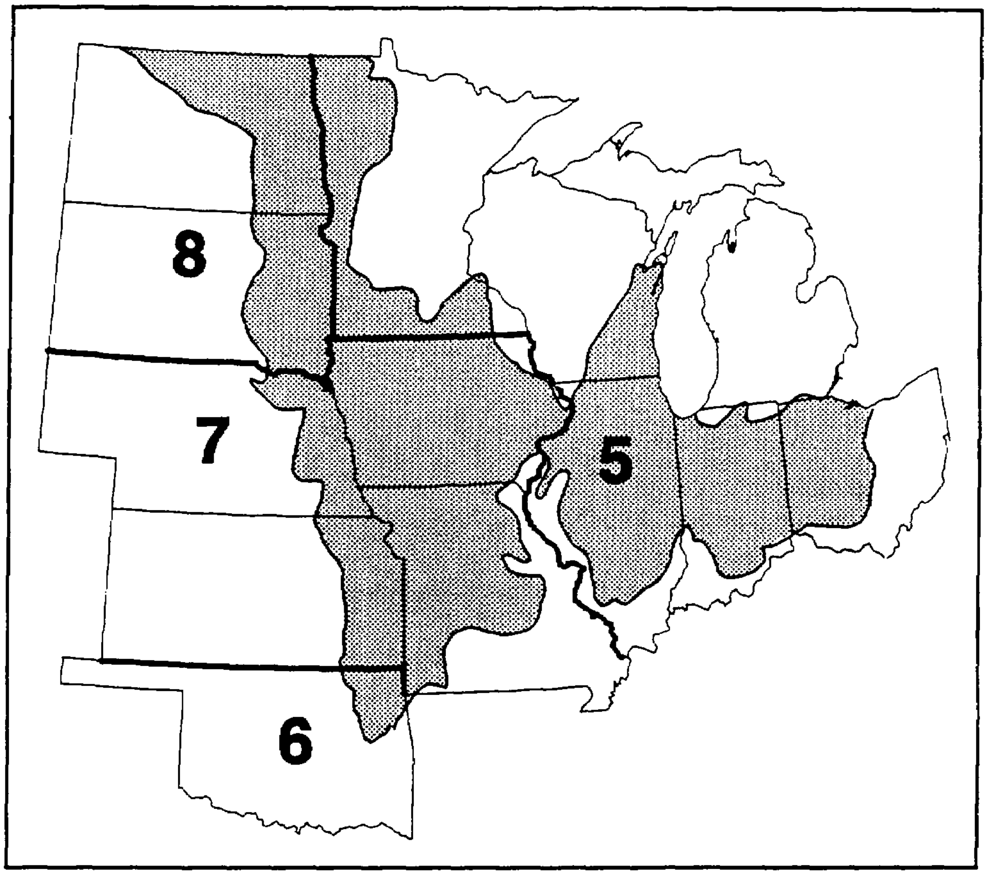
Contacts and Information Sources

When considering habitat conservation issues in an environmental impact assessment for the *Northern Lakes and Forests*, the reviewer should consult the following organizations and individuals for information on habitat impacts and mitigations:

State Natural Heritage Programs
U.S. Fish and Wildlife Service, Regional and Area Offices
State Fish and Game Departments
University and Research Programs
Herbaria and Museums

Lesley Sneddon, Regional Ecologist, The Nature Conservancy
Ralph Pisapia, Associate Director, Fish and Wildlife Enhancement, U.S. Fish and Wildlife Service

Midwest Habitat Region: *Midwest Croplands*



Geographical Description of the Region

The Habitat Region, *Midwest Croplands*, contains parts of 13 states. The region includes parts of Ohio, Indiana, Illinois, Missouri, Iowa, Wisconsin, Minnesota, North Dakota, South Dakota, Iowa, Nebraska, Kansas, and Oklahoma. Parts of EPA Regions 5, 6, 7, and 8 are included. The accompanying map indicates the boundaries of this habitat region and the states it comprises.

The *Midwest Croplands* comprise eight ecoregions (Omernik 1987). The vegetation of the *Midwest Croplands* includes a range of mosaic of bluestem, prairie (bluestem and indiangrass), oak, hickory, wheatgrass, needlestem, oak savanna, maple, basswood, beech, elm, and ash. The land use patterns are croplands and croplands with grazing lands.

Habitats of Concern

The *Midwest Croplands* contains many habitats of concern; the most obvious fall into the four general categories of oak savannas, native prairie remnants, wetlands, and old-growth central hardwood forest. The principal habitats of concern most at risk in the *Midwest Croplands* are listed below.

PRINCIPAL HABITATS OF CONCERN
IN THE *MIDWEST CROPLANDS*

1. Oak savannas

2. Native prairie remnants
   - tallgrass
   - little bluestem prairie
   - hill prairie

3. Wetlands
   - bottomland hardwoods of the Mississippi and Platte Rivers
   - prairie potholes
   - riparian corridors (only a few remaining)

4. Remnant central hardwood forest (virtually none left)

Habitat Values and Trends

The Corn Belt States of the Midwest have sustained the greatest conversion of terrestrial environments to human land uses in the nation. The elm-ash forest type in Ohio, Indiana, and Michigan has been reduced by 88% as a result of conversion to agricultural and urban uses (Klopatek et al. 1979). Bluestem prairie and its transition zone with oak-hickory forest has declined by 85% and 78%, respectively, representing a loss of more than 41 million ha, primarily due to conversion to agriculture. The agricultural states of Iowa, Illinois, and Indiana have lost the highest amounts of their natural ecosystems (92, 89, and 82%, respectively).

As with forest habitats, the spatial pattern and fragmentation of prairie vegetation can negatively affect native fauna and ecosystem health. The loss of grassland habitat to agriculture is responsible for the decline in prairie birds, especially those requiring large continuous habitats, and is analogous to the reduction in old-growth forests and its obligate species. The upland sandpiper, bobolink, dickcissel, grasshopper sparrow, savannah sparrow, and Henslow's sparrow all declined by 90% between the 1950s and 1970s (Graber and Graber 1983). Based on 1984 maps (USDA Forest Service 1989), the average number of endangered and threatened species per county is 2.4 for the Midwest Habitat Region, the lowest in the nation. Many historical species, however, have been extirpated from the Midwest.

Therefore, the few remaining natural areas are the major contributors to the diversity of the region. These areas include isolated examples of savanna, grasslands, and forests.

Savanna

Oak savanna once covered between 11 and 13 million ha of the Midwest in the states of Minnesota, Iowa, Missouri, Illinois, Wisconsin, Michigan, Indiana, and Ohio (Nuzzo 1986). It is now the rarest major habitat type in the Midwest; in 1985, only 113 sites totaling 2,607 ha of high-quality oak savanna remained in the Midwest, representing 0.02% of its original extent. Oak savanna is dominated by oaks producing 10 to 80% canopy, with or without a shrub layer, and has a herbaceous, predominantly grassy ground layer of prairie or forest species. Because savanna is fire-dependent, it rapidly converts to forest without fire or severe droughts. This occurred over much of its range within 40 years of settlement. Fire was eliminated by plowing and grazing, and by the construction of roads and railroads, which act as firebreaks. Other than a few areas with the appropriate moderate grazing or occasional fires, existing savanna occurs only on droughty sandy or rocky soils.

Grasslands

Prairie habitats constitute another important regional habitat that is greatly reduced in area. Only minor remnants of the vast area of tallgrass prairie remain. Restoration activities, a major component of prairie conservation efforts, have been attempted (1) by upgrading existing degraded prairies, and (2) by establishing prairie communities on sites without existing prairie species (Kline and Howell 1987). In addition to planting and site preparation techniques, fire is an essential tool in prairie restoration. Unfortunately, most restored prairies contain unwanted species and require special management involving site preparation and fire to address exotic herbs and woody species, respectively.

Forests

Merritt (1980) described the forests of the central region of the United States as comprising 40 million ha of the originally greater than 140 million ha of hardwood forest, or about 15% of the total land area [however much of these forests occur outside the Midwest Habitat Region in the states of Kentucky and Arkansas]. These forests have a long history of disturbance from Indian and European slash and burn systems, plus livestock grazing and logging. Throughout the Midwest, both the hilly well-drained soils and the more fertile wetter, glaciated areas have been cleared for agriculture. Woodlands not cleared for farming were heavily timbered. By the 1930s, permanent clearing had created the most fragmented forest system in the United States.

The most extensive forest type, oak-hickory, makes up 72% of the forest acreage, while elm-ash-cottonwood occupies about 17%. Today, woodlands are limited in size, are widely dispersed, and occur primarily in the portions of the land that cannot be easily worked for row crops. Along the prairie fringes, wooded areas are located on steep bluffs and ravines and along poorly drained bottomlands. Elsewhere, they are found on rough and rocky land, on poorly drained uplands, along stream banks, and on bottomlands subject to overflow. These few remaining forests are especially important because of their role as riparian areas in the ecological functioning of the watershed. Nationwide 70% to 90% of riparian areas have been lost to human activities (Ohmart and Anderson 1986).

Even in the last 25 years, total Midwest forest has continued to decline. Only 100,000 ha or 0.07% of the original central hardwood old growth remains, mostly in protected areas that were once family farms. The long-term viability of this forest type is in question due, in part, to the "natural" change from oak-hickory to sugar maple (perhaps from reduced fire or climate change) occurring on mesic sites. Degradation is continuing from recreational overuse and vandalism, and from adjacent impacts such as urban construction, soil erosion, agricultural chemicals, land drainage, and strip mining for minerals (Parker 1989).

Wetlands

Prairie wetlands, located in the glaciated portion of the states of North Dakota, South Dakota, Minnesota, and Iowa, constitute the single most important breeding area for waterfowl in North America (Hubbard 1988). These wetlands support 50% to 80% of the continent's duck populations as well as many other wildlife species such as nongame birds, muskrat, and mink. These wetlands, or prairie potholes, are relatively shallow, water-holding depressions varying in size, water permanence, and water chemistry. Refilling usually occurs from spring precipitation and runoff, and water levels fluctuate widely due to climate variability (Poiani and Johnson 1991).

Other wetlands include diverse shallow wetlands, ponds, and lakes that were glacially formed, and bottomland hardwoods. The peak loss in bottomland hardwood habitat occurred in the 1970s and 1980s, and losses have declined since then for economic reasons.

Activities and Impacts Affecting Habitats

The following activities result in major impacts on habitats of concern in the *Midwest Croplands*:

- Conversion to agriculture and offsite impacts of cultivation practices (especially to aquatic systems).
- Urban development, both residential and commercial (particularly in large metropolitan areas such as Chicago, St. Louis, Cleveland, and Minneapolis-St. Paul).
- Forest loss and fragmentation (especially to highway development and channelization of riparian areas).

The major impacts of degrading activities on the principal habitats of concern are summarized in the table below.

IMPACTS ON HABITATS OF CONCERN IN THE *MIDWEST CROPLANDS*					
	Land Conversion	Timber Harvesting	Grazing	Water Management	Other
Oak savanna	Major conversion to agriculture	Moderate	Moderate	Minor	Succession to forest after fire suppression
Prairie	Major conversion to agriculture	None	Major	Minor	Invasion of exotic species
Wetlands	Major conversion to agriculture and urban uses	None	None	Drainage for land conversion	Minor
Remnant forests	Conversion to agriculture	Removed prior to agriculture	None	None	Minor

Land Conversion

Historically in the Midwest, conversion to agriculture has been a major factor affecting habitat loss. In Illinois and Indiana more than 80% of the natural ecosystems have been lost to agriculture. Conversion to agriculture is continuing on the fence rows and ditch banks that remain. Odd-dimensioned plots are now being converted as a result of monetary incentives in Wisconsin and other states (Todd Peterson, personal communication). During the Illinois state inventory of prairies, lands were disturbed for railroad maintenance or converted to agricultural fields faster than they could be identified. These conversions represent the loss of the only remaining wildlife habitats in many areas (Illinois Department of Conservation 1978). This is especially true of bottomland hardwoods, which were also affected by channelization and timber harvesting. Logging continues on the last large tracts of forest, including accelerated development via barge canal along the lower Kaskaskia River (the largest remaining tract of bottomland timber in Illinois). The loss of riparian areas has resulted in declines among the waterfowl of the Mississippi Flyway.

Approximately 60% of North Dakota's original 5 million ac of prairie pothole wetlands has been lost (Stromstad and Donovan, 1989). Agricultural development accounts for nearly 99% of prairie pothole losses. In northeastern Illinois, 20% of wetlands identified by aerial photos were filled for construction between 1970 and 1974. Instances of new wetland drainage appear to have dropped significantly; however, upland grasslands adjacent to wetlands are still significantly at risk. Approximately 50% of the grasslands in the Missouri Coteau of North Dakota were converted to cropland between 1965 and 1975. Loss of grasslands, hayed and grazed for livestock production, adversely affect many species, including the elimination of upland nesting cover for ducks. Some limestone glades are being quarried; hill prairies are being used for homesites; railroad prairies face new maintenance threats from herbicides and heavy machinery; and new lands are now being cultivated.

Agricultural Impacts

Both the extensive coverage and intensive use of agricultural land in the Midwest pose additional stresses to habitats through cultivation practices (NRC 1982). The use of fertilizers and pesticides, irrigation and drainage, double cropping and increased field size all contribute to increased pollutant loads and severe impacts on habitats. Agricultural chemicals are toxic to many species and can negatively affect population levels, community composition, and ecosystem dynamics. Other intensive cultivation practices directly reduce important hedgerow and riparian habitat and usually produce severe offsite impacts.

Impacts on Aquatic Systems

The intensive use of midwestern lands converted to human uses has resulted in a high level of pollution discharge and other negative impacts on aquatic systems. A historical example is the degradation of the Illinois River through intensive human use from Chicago, including sewage discharge, dredging, damming, barge traffic, and introduction of carp. As a result, half of the original 400,000 ac were drained, and the other half of the sand-bottom backwaters of the river were covered with mud.

Smaller streams throughout the Midwest have also been severely degraded through the impacts of agricultural practices and urban expansion. In particular, fish populations have been extirpated by the following factors (in order of relative importance):

- Siltation.
- Drainage of wetlands.
- Stream desiccation due to lowered water tables.
- Competition and hybridization due to habitat changes and introduction of exotic species.
- Pollution.
- Dams and impoundments.
- Raised water temperatures with removal of streamside vegetation.

Mitigation of Impacts

The conservation of habitats requires consideration of mitigations for the major activities impacting habitats of concern. In the Midwest Habitat Region, the primary habitat impacts are due to the following:

- Conversion to agriculture and offsite impacts of cultivation practices.
- Urban development, both residential and commercial.
- Forest loss and fragmentation.

In the Midwest, habitat conservation of oak savannas and prairie types is essentially a restoration and creation effort. Less habitat of high ecological integrity remains in the Midwest than in any other region except the central valley of California and parts of Florida (Steve Chaplin, TNC, personal communication). Restoration of grassland systems concentrates on revegetation and borrows largely from agriculture and horticulture (Jordan et al. 1988). The most commonly measured parameters at restoration sites are the survival and growth of planted vegetation for the first few growing seasons, generally too short a period to evaluate the ultimate species diversity or the presence of self-regeneration. More successful has been the use of a "prairie matrix" (developed by Robert F. Betz) of a few aggressive and tolerant native species that survive weed competition too intense for many other native plants (Packard 1988). Restorationists follow this matrix with less aggressive species to effectively shorten the natural ecological succession of prairies.

Degradation of remnant forest is continuing from recreational overuse and vandalism and from adjacent land-use practices such as urban construction, soil erosion, agricultural chemicals, land drainage, and strip mining for minerals. Research is needed to determine whether important mitigation factors (e.g., adjacent harvest, increased access through new roads, different harvest systems, and width of buffers) can be applied (Parker 1989).

Land Conversion

Effective mitigation of land conversion activities can sometimes be obtained only by avoiding impacts on rare or unusual habitat types. Rarely, if ever, is restoration or compensation an adequate mitigation for the loss of these habitats. In these cases, mitigation is a siting issue, where construction and degrading activities are located at a distance from the habitats of concern. The habitat is adequately preserved if all possible impact scenarios are accounted for. Barring this solution, effective management measures must be implemented to ensure the protection of the habitats of concern.

In the case of unique riparian or wetland habitats, hydrological and contamination concerns are especially important. Construction or resource management activities require the use of sediment filter strips and other means of intercepting offsite contaminants. Road building and structural "improvements" must not result in altered hydrological regimes. Where rare plant types exist or where habitats are unstable, recreational access may have to be limited. These mitigations can be best implemented by creation of a regional land-use plan (through a coordinating council like the Waterfowl Flyway Council) and landowner incentives (like the Conservation Reserve Program).

Conversion to agricultural land is a special concern in the Midwest. Land conversion to agriculture can cause ground water overdraft, salinization of topsoil and water, reduction of surface water, high soil erosion, and destruction of native vegetation. Mitigations include more conservative irrigation techniques and improved drainage systems. Soil conservation techniques vary from windbreaks to contour plowing, stripcropping, rotation of crops, conversion to grass, and/or minimum tillage.

Agricultural Impacts

Maintenance of riparian areas and habitat corridors is effective mitigation for intensive agriculture. Implementation of integrated pest management (IPM) practices can reduce the load of toxic agricultural chemicals entering both terrestrial and aquatic systems. In general, institution of best management practices (BMPs) that address nonpoint source pollution are appropriate mitigations for impacts caused by cultivation practices.

Wetlands

Mitigation of wetlands destruction and degradation is the subject of a growing body of literature (Kusler and Kentula 1989). Restoration and mitigation banking concepts are still being evaluated as effective mitigation measures for direct wetlands alterations.

Guidelines for Reviewers

Reviewers of environmental impact assessments will find this document useful if they follow the steps laid out in the introduction:

1. Review the status and trends of habitats in the region.

2. Identify the habitats of concern.

3. Link the activities involved with impacts to these habitats of concern.

4. Devise appropriate mitigations for the impacts.

Each reviewer can then determine the adequacy of the environmental impact assessment in question and recommend modifications to enhance its effectiveness.

In identifying the habitats of concern, the reviewer should supplement the information in this document with detailed locational information on the abundance and distribution of habitats within the region of interest, and with any historical information on the extent and quality of these habitats. Most important, the reviewer should characterize the habitats in terms of their ecological values (e.g., use of wooded wetlands by migratory waterfowl).

In considering the links between activities and habitats, the reviewer should look beyond direct impacts to indirect and subtle effects, including cumulative impacts, interactive and synergistic impacts, and scale-dependent impacts (e.g., effects of fragmentation on ecosystem integrity and species home ranges).

In devising possible mitigations, the reviewer should follow the seven principles for habitat mitigation repeated below. The reviewer should also determine whether adequate assurances have been given that the mitigations proposed will be completed.

1. Base mitigation goals and objectives on a landscape-scale analysis that considers the needs of the region.

2. Mimic natural processes and promote native species.

3. Protect rare and ecologically important species and communities.

4. Minimize fragmentation of habitat and promote connectivity of natural areas.

5. Maintain structural diversity of habitats and species diversity, where appropriate, to promote the natural variety of the area.

6. Tailor management to site-specific environmental conditions and to the unique impacts of the specific degrading activity.

7. Monitor for habitat impacts and revise mitigation plans as necessary.

Finally, the reviewer should consider the proposed activities and mitigations in the context of relevant regional program goals and objectives (e.g., whether the outcome of the project will be in accordance with principles set out by regional planning commissions).

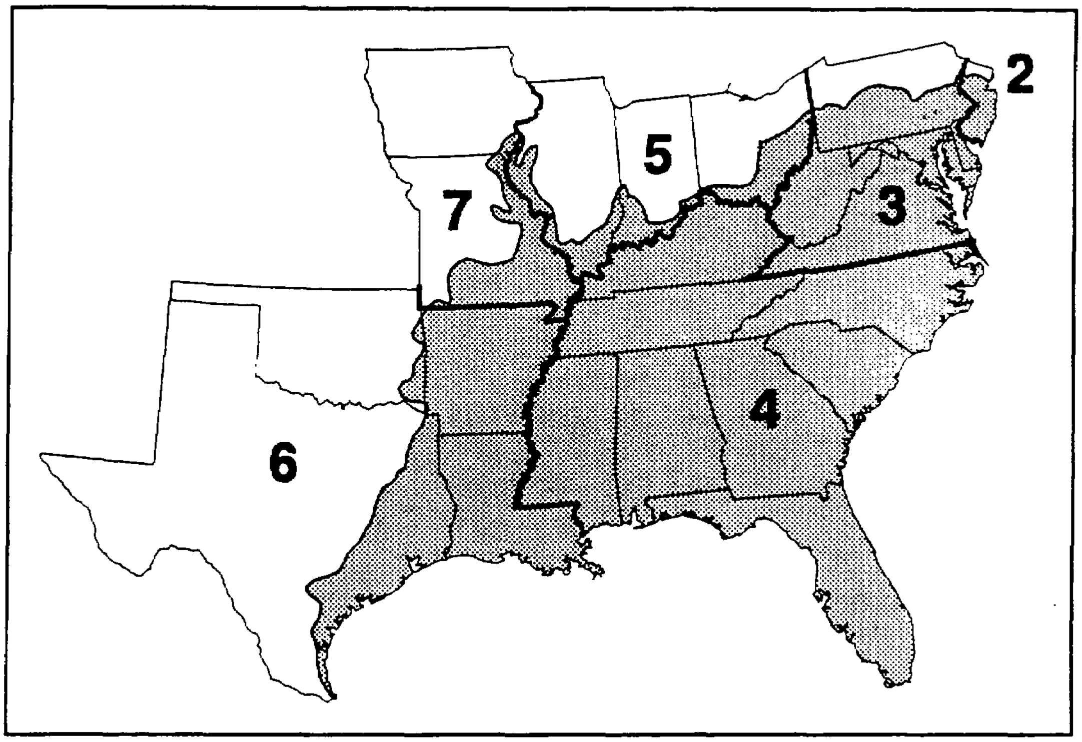
Contacts and Information Sources

When considering habitat conservation issues in an environmental impact assessment for the *Midwest Croplands*, the reviewer should consult the following organizations and individuals for information on habitat impacts and mitigations:

State Natural Heritage Programs
U.S. Fish and Wildlife Service, Regional and Area Offices
State Fish and Game Departments
University and Research Programs
Herbaria and Museums

Steve Chaplin, Regional Zoologist, The Nature Conservancy
Mamie Parker, Division of Federal Activities, U.S. Fish and Wildlife Service Region 3

Southeast Habitat Region: *Southeastern Forests and Croplands*



Geographical Description of the Region

The Southeast Habitat Region, *Southeastern Forests and Croplands*, contains all of 14 states (and the District of Columbia) and parts of 9 states. The region includes all of Maryland, Delaware, District of Columbia, Virginia, West Virginia, Kentucky, Tennessee, North Carolina, South Carolina, Georgia, Alabama, Mississippi, Florida, Louisiana, Arkansas, and parts of New Jersey, Pennsylvania, Ohio, Indiana, Illinois, Iowa, Missouri, Oklahoma, and Texas. EPA Region 4 is included in its entirety, and parts of EPA Regions 2, 3, 5, 6, and 7 also are included. The accompanying map indicates the boundaries of this habitat region and the states it comprises.

The *Southeastern Forests and Croplands* is perhaps the most diverse in the nation comprising 20 ecoregions (Omernik 1987). The vegetation of the region includes a wide range of forest types, including Appalachian oak, oak/hickory/pine, mixed mesophytic forest, southern mixed forest, southern floodplain forest, as well as palmetto prairie and everglades. Northern hardwoods and southern mixed and floodplain forests are also present. The land use pattern is mostly a mosaic of forest and cropland with substantial woodland, pasture, swampland, marshland, and urban components.

Habitats of Concern

The *Southern Forests and Croplands* contains many habitats of concern; the most obvious fall into eight general categories. The principal habitats of concern most at risk in the *Southern Forests and Croplands* are listed below.

PRINCIPAL HABITATS OF CONCERN
IN THE *SOUTHEASTERN FORESTS AND CROPLANDS*

1. Bottomland hardwoods
   oak/gum/cypress

2. Scrub habitat
   Florida sandpine scrub
   southeastern savanna and bogs

3. Spruce-fir forest in the Appalachians

4. Old-growth pine forest
   longleaf pine-wiregrass
   New Jersey pine barrens

5. Everglades ecosystem
   other wetlands

6. Maritime forest of coastal barriers

7. Contiguous upland hardwood forests

8. Mature mixed mesophytic Appalachian and Ozark forest
   subalpine, barrens, and caves
   old-growth forest

Ecosystems of concern include the Chesapeake Bay and major river systems, abundant freshwater and coastal wetlands, relict closed boreal subalpine forest communities, limestone barrens, remnant alpine peat bogs, and the Great Dismal Swamp. Also, the endemic communities in the Southern Appalachians, high-elevation spruce-fir forests (boreal subalpine), bottomland hardwood forests, coastal live oak forests, long-leaf pine wiregrass hardwood hammocks, and the Everglades. Oak-gum-cypress forests of the lower Mississippi drainage are important overwintering habitats for avian species.

Habitat Values and Trends

Two to three centuries ago, almost all of the land area in the South was forested. Since that time, agricultural land has become an increasingly prominent part of the landscape (USDA Forest Service 1989). The loss of forested area accelerated in the late 1800s with the harvesting of old-growth forests. However, after 1920 forest area began to increase with the abandonment of agricultural land, reduced timber harvesting, and efforts to regenerate forests. This trend continued until the 1960s, when abandonment slowed and new clearing for agriculture and pastureland began (at first among bottomland hardwoods and later more uniformly across the South). Concomitant increases in population and industry saw large areas converted to residential and commercial uses. Future economic conditions will likely determine whether high rates of conversion continue. Projections for the next 50 years show urban area increases of 14 million ac leading to losses of several million ac each in cropland and pastureland. These losses may stimulate forest conversions for additional agricultural land; in particular, forests of the Ozarks are expected to be converted to pasture (NRC 1982).

<u>Forests</u>

The Southeast contains 200 million ac of forest land with 62 million ac in pine forest (loblolly-shortleaf pine, longleaf-slash pine, and oak-pine), 71 million ac in oak-hickory, and 31 million ac in bottomland hardwood types (USDA Forest Service 1989). Since 1963, losses in the Southeast have occurred in longleaf-slash pine (40%), oak-gum-cypress (24%), and loblolly-shortleaf pine (15%). Bottomland hardwoods have been lost to agricultural clearing, and most remain only as strips along streams where the soil is too wet for cropping or grazing. They are further endangered by dams and drainage modifications. The loss of longleaf pine habitat can be attributed to the logging of nearly all original forest from the Atlantic coast to the Piney Woods of Texas and the replacement with loblolly and slash pine. Losses of other pine species are the result of poor pine regeneration and less farmland abandonment.

The forests of the *Southeastern Forests and Croplands* contain a particularly diverse fauna and flora. Many northern species complexes reach their most southern extent in the southern Appalachians, while many southern species reach their most northern extent at Cape Hatteras. Based on 1984 maps (Flather and Hoekstra 1989), the average number of endangered and threatened species per county is 5.7 for the *Southeastern Forests and Croplands*. The following listing of southeastern forest types illustrates some characteristic ecological values of the region:

- Loblolly-shortleaf pine - much of the ecosystem has been converted to pine plantations, often mixed with pasture or row crops.

- Longleaf-slash pine - covers the coastal region and has an extensive grassy understory that varies with site and geographic location; it supports many endemic plants and endangered animals including red-cockaded woodpecker and Florida panther; nearly eradicated in logging boom of the early 1900s, it was replanted in loblolly or shortleaf pines; slash pine now dominates this ecosystem.

- Oak pine - often occurs on cutover sites with poor pine regeneration; supports white-tailed deer and wild turkey.

- Oak hickory - supports southern bald eagle, red wolf, red-cockaded woodpecker; is widespread with at least six distinct associations.

- Bottomland hardwood - principally oak-gum-cypress and elm-ash-cottonwood ecosystems; mangrove swamps in Florida support Florida manatee, brown pelican, bald eagle, hawksbill sea turtle, and Atlantic Ridley sea turtle; cypress savanna has been mostly converted to pasture and cropland, but remaining areas support fox squirrel, ibises, cormorants, herons, egrets, kingfishers, Bachman's warbler, Florida panther, and bald eagle; elm-ash-cottonwood supports many waterfowl species.

Although the logging of mature forests may increase site diversity by creating forest edge, these timber harvesting activities usually increase the number of species that are not in need of protection (e.g., white-tailed deer, bobwhite quail, cottontail rabbit, gray squirrel, wild turkey) at the expense of species that are regionally, as well as locally, rare or vulnerable. As is the case with old-growth Douglas fir in the Northwest, the decline of subalpine Appalachian forests threatens the last remnants of historical ecosystems, the loss of which would dramatically lower regional and global diversity.

Of particular concern in the Southeast are (1) old-growth-dependent species (such as the red-cockaded woodpecker in coastal plain pine forests), and (2) forest-interior-dependent species (including many neotropical migrant songbirds in mixed deciduous forests). Because of their ecological complexity and relative isolation, southeastern forest ecosystems contain many rare and endangered species that require mature trees for nesting and foraging. Mature trees are at serious risk from logging in the *Southeastern Forests and Croplands*; though sustainable short-rotation plantation forestry dominates the region, remaining areas of mature forest are still being sought and exploited for short-term profits.

The habitat of the red-cockaded woodpecker, which exists in the southern pine forests ranging from Maryland to Texas, has been reduced and will continue to decline under current timber harvesting management practices (Roise et al. 1990). The causal factor in habitat loss is the cutting of loblolly pine-dominated stands greater than 75 years of age, and the cutting of all longleaf pine stands greater than 95 years of age. Lennartz et al. (1983) estimates that pines required by the red-cockaded woodpecker have declined by 13% in 25 years. In Texas, clear-cut logging has been restricted because of concerns for the red-cockaded woodpecker (Larmer 1989).

Species described as interior forest birds (Terborgh 1989) are of special concern in forest environments suffering from fragmentation. Songbirds, in particular, are declining in number because of the loss and fragmentation of forest habitat along their migratory path from New Hampshire to Mexico. Forest conversion and fragmentation leads to an increased likelihood of starvation and an increased likelihood of predation due to an increase in the numbers of songbird predators (Terborgh 1974). Robbins et al. (1989) summarized the breeding habitat losses and requirements of forest birds of the Middle Atlantic States in light of the negative effects occurring from forest fragmentation (due to suburban expansion) in that region (Lynch and Whitcomb 1978). They concluded that in relatively undisturbed mature forests, the degree of isolation and the area of forest were better predictors of relative abundance of bird species than were any habitat variables. Forest reserves of thousands of hectares are required to have the highest probability of providing for the least common species of forest birds in a region.

Bottomland hardwood forests represent a third important forest habitat of the Southeast, one that supports many bird species during the critical over-wintering period. The oak-gum-cypress ecosystem of the southern states also includes a diverse resident avifauna (Dickson 1988). For example, this habitat was the historical range of the ivory-billed woodpecker. The forests of the Appalachian and the Ozark regions also contain valuable habitats.

The Appalachian Plateau has special value because its cool, wet climate at 2,400 ft allows northern species to live at lower latitude. Encompassing more than 230 terrestrial vertebrate species, this region has the richest floral, breeding bird, mammal, and amphibian communities of any upland eastern U.S. forest type (Hinkle et al. 1989). More than 60% of the breeding birds are neotropical migrants. The mature mixed mesophytic forest contains many old-growth areas and unique habitats such as subalpine, montane grasslands, serpentine areas, shale barrens, mountain peatlands (supporting unusual plants and animals: larch, wild calla, cotton grass and northern water shrew), vernal ponds (rare amphibians and invertebrates), sandstone ridgecrests (rare plants), and caves (globally rare aquatic amphipods) (The Nature Conservancy, Maryland Chapter 1991.)

The forests of the Ozark region (encompassing southern Missouri and northern Arkansas) were once vast tracts of white oaks and shortleaf pines, but today they exist as a mosaic of relatively young vegetation in various stages of succession (Smith and Petit 1988). At the turn of the century, the region experienced perhaps the most extensive destruction of forest through clearcutting on the continent. This resulted in the loss of many bird species dependent on mature forest and the increase of species adapted to open environments. Of the forest birds that have survived the transformation to a mosaic of young forest, the broad-winged hawk and hooded warbler are at risk from increased habitat fragmentation and conversion of hardwood forest to pine plantations.

Grasslands, Barrens, and Scrub Habitats

In addition to mature forests, terrestrial habitats of significance in Maryland and other mid-Atlantic states are shale barrens, barrier islands, serpentine areas (rock outcrops), peat lands, floodplains, and sandstone glades. Serpentine sites represent the kind of unusual local environments that produce unique habitats throughout the region. A relatively high percentage of vascular species on state natural heritage program lists and on the candidate lists of threatened and endangered species of the United States are serpentine endemics. Currently, more than 400 communities are listed in the Maryland Natural Heritage Program database with another 200 species having been extirpated (Janet McKegg, Maryland Natural Heritage Program, personal communication). Many other important communities are aquatic or riparian (e.g., the Delmarva bays), but are often better protected by federal and state wetlands regulations. The New Jersey Pine Barrens is another region with many important local habitats. This pinelands ecosystem comprises a mosaic of upland, aquatic, and wetland environments covering more than 400,000 ha (McKenzie 1981).

Sandpine scrub is one of the nation's most threatened habitats; it is found only on scattered knolls of coastal and inland Florida and adjoining Alabama and Georgia (Bass 1988). It has been reduced to one-fifth of its original acreage by expanding agriculture and industry. Along with mahogany hammock, sandpine scrub is also the least recoverable of habitats in Florida. It has perhaps the highest concentration of endemic plants (including many that are endangered or threatened) of any place in the United States. Development is the principal threat, and landowners are bulldozing areas to prevent federal protection of undisturbed scrub. The scrub is already vulnerable because the natural burn cycle of the scrub has been disrupted by fire suppression practices. This vegetation type requires burn cycles of 30 to 80 years to allow dominants to reproduce but at the same time to prevent canopy closure.

Savannas and bogs of the southeastern coastal plains are also habitats sensitive to fire management. Without fire they are invaded by fire-intolerant trees. These ecosystems are home to many endemics, such as carnivorous plants. Approximately 97% of southeastern savannas and bogs no longer exist, having been converted to pine plantations or pastures through drainage or to farm ponds in hillside bogs.

The only substantial rangelands in the Southeast (4 million ac or 13% of the total area) are the wet prairies and marshes along Atlantic and Gulf coasts that include the Everglades and palmettos prairie of southern Florida. Louisiana and Texas also possess significant portions of this ecosystem; unique species include the golden-cheeked warbler, Texas red wolf, Attwater's prairie chicken, Florida panther, Florida great white heron, Everglades kite, plus the more common collared peccary, coatimundi, and pronghorn antelope. Species of concern include subtropical natives suffering population declines due to the loss of habitat and invasion by exotic species. The region also contains many freshwater and marine habitats, and is unique in the number and diversity of its wetland habitats.

Wetlands

Specific southeastern wetland problem areas identified by Tiner (1984) include the following:

- Estuarine wetlands of the U.S. Coastal Zone.
- Louisiana's coastal marshes.
- Chesapeake Bay's submergent aquatic beds.
- South Florida's palustrine wetlands.
- Forested wetlands of the Lower Mississippi Alluvial Plain.
- North Carolina's pocosins.

In the Southeast, 86% of the forested wetlands are in the coastal plain (Tansey and Cost 1990). In the last 10 years, 16% of the area has been converted to nonwetlands through changes in species or hydrology, including harvesting. Large losses of forested wetlands in the Lower Mississippi Valley have occurred with the conversion of bottomland hardwood forests to cropland. Of the 11.8 million ac of bottomland hardwood forest in 1937, only 5.2 million ac remain, including 60% in seasonally flooded basins or flats and 40% in wooded and shrub swamps. These decreases in acreage were matched by increases in croplands, principally soybeans, and corresponded to the completion of major Corps of Engineers flood control projects and smaller watershed projects. Indirect effects of these projects (clearing by landowners in anticipation of flood protection) exceeded losses to direct construction. The rate of loss continues to increase in Louisiana.

Shrub wetland losses are greatest in North Carolina owing to the conversion of pocosins to cropland and pine plantations and their mining for peat. The drainage of inland marshes is greatest for the Florida Everglades. Indeed, the modifications to the water drainage patterns beginning in the headwaters of Kissimmee basin through Lake Okeechobee to the Everglades are some of the most extensive in the country. Additional losses of mucky bottomlands, marshes, and dunes across the coastal plain have decreased duck populations, flood control, and water supply. More than 50% of Texas wetlands (including bottomland hardwoods and coastal marshes) have been lost (Loftis 1991). In the Gulf

Prairies and Marshes region, much of cordgrass marshes are drained and barrier islands overgrazed resulting in severe soil erosion.

Aquatic Systems

Approximately 24 million ac of water area are contained in the lower Mississippi River and tributaries, the lakes and waterways of the Mississippi Delta, the large number of small and large lakes in Florida, the numerous large water impoundments, the small ponds and streams, and the Atlantic and Gulf coastal waters (one-fifth of this area) of the *Southeastern Forests and Croplands*. The many unique aquatic habitats make this region the most diverse in the nation.

The marine systems in the Southeast are exceptional and include the unique coral reefs of Florida. A 5-year study on the Florida Keys coral reef by University of Georgia and Florida Institute of Oceanography indicates a 10% per year decline in some parts of the reef and predicts possibly irreversible endangerment in the next decade (Keating 1991). Threats include pollution (especially nutrients from sewage and agriculture that stimulate algae overgrowing), sedimentation (from erosion via forest and shoreline conversion that smothers corals), diseases (possibly aggravated by water quality stresses), and weather (including global warming).

Activities and Impacts Affecting Habitats

The major sources impacting habitats in the *Southeastern Forests and Croplands* include residential, industrial, and commercial developments, logging and silviculture practices, agricultural activities, mining practices, and interstate highway or expressway construction. These activities have produced adverse impacts on species populations and their behavior, as well as on ecosystem processes such as energy flow and nutrient cycling. They have also contributed to the proliferation of nuisance plants and animals. In its comparative risk analysis for the mid-Atlantic states, EPA Region 3 (1988) ranked adverse effects on ecosystems as high from silviculture, coal mining, and conversion to urban uses through residential construction; as moderate from agriculture, mineral mining, second homes development, dam construction, and recreation; and as low from oil and gas development, bridge construction, and water use. In the more southern states, timber harvesting and agriculture have even greater impacts on habitats. In the Gulf Coast States, oil and gas production is a major activity degrading coastal environments.

The following activities result in the major impacts on habitats of concern in the *Southeast Forests and Croplands*.

IMPACTS ON HABITATS OF CONCERN IN THE *SOUTHEASTERN FORESTS AND CROPLANDs*					
	Land Conversion	Timber Harvesting	Mining	Water Management	Other
Bottomland hardwoods	Conversion to agriculture	Moderate	None	Impacts of impoundments and siltation dams	Minor
Scrub habitat	Conversion and fragmentation for residential development	None	None	Minor	Minor
Spruce-fir forest	Conversion for resort development	Moderate	None	None	Acidification
Old-growth pine forest	Moderate	Conversion to short rotation plantations	None	Minor	Military activities
Everglades and wetlands	Urban and agricultural conversion	Minor	Peat mining of pocosin wetlands	Impacts of water diversions altering hydrology	Invasion of exotic species
Maritime habitats	Conversion for coastal development	Minor	None	Minor	Recreational activities
Contiguous forest	Fragmentation from urban sprawl and highway development	Fragmentation	Fragmentation	Minor	Minor
Appalachian and Ozark forests	Conversion to plantation silviculture	Major	Major impacts of coal mining	Minor	Minor

Land Conversion

Historically, land conversion of both uplands and wetlands has profoundly affected the natural communities in the Southeast. In recent years, the boom of population growth has caused increased conversion of natural areas to industrial and residential development. Rapidly growing areas in Florida and certain SunBelt cities are suffering intense "spin-off development" associated with highway development, a process that is rapidly expanding into previously rural areas. This increased road construction is causing severe fragmentation of sensitive environments such as the North Carolina pocosins and the Florida sandpine scrub. The sum of this massive habitat alteration in areas such as south Florida has been a dramatic reduction in not only large mammals and birds but also reptiles and amphibians (Crowder 1974). Conversion of bottomland hardwoods to agriculture continues to be a significant cause of habitat loss that has detrimental effects on the waterfowl of the Mississippi Flyway.

Agricultural and Grazing Impacts

The use of agricultural land in the Southeast poses additional stresses to habitats through cultivation practices (NRC 1982). The use of fertilizers and pesticides, irrigation and drainage, double cropping, and increased field size all contribute to increased pollutant loads and severe impacts on habitats. Agricultural chemicals are toxic to many species and can negatively affect population levels, community composition, and ecosystem dynamics. Other intensive cultivation practices directly reduce important hedgerow and riparian habitat and usually produce severe offsite impacts. Grazing has a lesser impact on the region as a whole but is increasing in the south Florida prairies and oak hammocks west of the Everglades.

Timber Harvesting

Timber harvesting activities are another major cause of habitat loss in the *Southeastern Forests and Croplands*, affecting many sensitive forest types. For example, the southeastern mixed forest and the Ozark forests are being converted to pine monocultures. Logging also continues in the southern Appalachian subalpine forest and some bottomland hardwood forests. These impacts affect 90% of the total bird, amphibian, and fish species and 80% of mammal and reptile species that utilize forest ecosystems (U.S. Forest Service 1989).

In addition to the direct destruction of forests through land conversion, timber harvesting activities can fragment, simplify, and degrade forest habitats. The faunal communities inhabiting forests vary with the successional, or seral, stage. Because the principal impact of timbering practices is to convert forest stands from latter to earlier seral stages, logging has a major impact on resident animal as well as plant species. Timber harvesting telescopes plant succession, shortens rotations, compresses seral stages, and decreases the proportion of old growth. The conversion of hardwoods to conifers creates structurally simplified plantations that reduce structural diversity and wildlife. This has produced a trend away from declining habitat types and toward common habitat types. Management for monotypic even-aged stands causes increases in forest pest damage that can result in large-scale spraying and the accompanying impacts. Logging activities also impact nearby aquatic systems through erosion and sediment transport.

Logging in the national forests of Texas relied exclusively on clearcutting and its variations until 1988. The general practice was to convert the natural complex forest systems (tall pines with oak, ash, and hickory underneath in diverse groves of 100 broadleaf tree and shrub species) into single-species loblolly pine plantations. Site preparation (including the clearing of all vegetation, concomitant removal of topsoil, application of herbicides, and burning) was conducted to eliminate competition with planted species. This homogenization threatened the long-term health and productivity of the forest by reducing the quality of the gene pool. Because of the susceptibility of monocultures to insect infestation, additional clearcutting was conducted to provide buffer areas around the pine plantations. Between 1978 and 1988, the number of colonies of the endangered red-cockaded woodpecker fell from 455 to 174. Recent court decisions and enlightened foresters are moving away from clearcutting and instituting selective timber harvesting in national forests containing the red-cockaded woodpecker: Texas, Louisiana, Alabama, Mississippi, South Carolina, North Carolina, and Kentucky (Larmer 1989).

Unlike the Pacific Northwest, little research has been conducted on mature eastern hardwood forest (virtually no old growth remains). However, results do show correlations between older forest and the abundance of several species, including great horned and barred owls, pileated and red-cockaded woodpeckers, and common ravens. Declines in other species have been attributed to brood parasitism (by brown-headed cowbirds) and nest predation (by common crows, striped skunks, opossums, black racers, and rat snakes) that occurs along clear-cut edges and in thinned stands. These edge effects are a prominent impact of forest fragmentation. Fragmentation is second only to the decrease in old-growth species as a major impact of timber harvesting activities. The faunal significance of this fragmentation includes discrimination against large-bodied species (e.g., Florida panther, red wolves, mink), genetic swamping by invading species, inbreeding through isolation of populations, and ecological release of middle-sized omnivores.

In addition, there has been the general decline in neotropical migrants that breed in eastern hardwood forests. Although the situation is complicated by losses of wintering habitats for long-distance migrants in Latin America, results indicate that species still present in large blocks of forest are absent from small patches (Robbins et al. 1989). Fragmentation of forest habitat from timber harvesting and from land conversions, especially for transportation, appears to be the major cause of these declines (Terborgh 1989) and has been especially severe in southeastern bottomland forests.

Mining

The greatest single threat to terrestrial habitat in West Virginia and Kentucky is coal mining, projected to increase from 2.4 million ac to 3.4 million ac (4% of total land area) by the year 2000 (McComb et al. 1991). The profitability of timber harvesting will be increased by the transportation infrastructure built for coal mining and the fact that large acreages have reached sawtimber age. This transition sets the stage for an unprecedented combination of cumulative impacts in the central Appalachians in the next 20 to 30 years. Surface mining will be conducted on ridge tops and side slopes; development of single-family housing will occur in valley bottoms; and mature hardwood will be harvested in midslopes and coves. In addition to the direct destruction of forests, the potential for severe soil erosion and offsite impacts is great.

Oil and gas extraction is important on the Gulf coast but rare in other parts of this region. Gold mining is currently causing habitat degradation in South Carolina.

Water Management

Historically, water management activities such as damming and diversion of rivers have had a major impact on the habitats of the Tennessee Valley, the Mississippi River floodplain, and other regions of the Southeast. For example, man's efforts to control the Mississippi River's flooding regime, enhance its navigation, and extract its minerals have led to a rapid deterioration of Louisiana's coastal environment. Wetland loss in Louisiana is more than 400,000 ac since 1900; only 45% of the original forested wetlands in Louisiana remain. The primary causal factor in this loss is subsidence of wetlands that are receiving inadequate amounts of sediment from the Mississippi. An accretion deficit results when levee systems and control structures transport sediments to deep Gulf waters.

In the Mississippi Basin (mid-south Alabama, Tennessee, eastern Texas, and Oklahoma), considerable acreages of bottomland hardwoods were lost to reservoir development between 1962 and

1985 (Gosselink and Lee 1987). Dam construction in general changes water flow patterns, causes flooding, and changes salinity patterns; this kills tree seedlings and can convert forest to salt marsh. Water diversion, another activity degrading southeastern habitats, is severely impacting the Everglades. This diversion stems from the competition for water by agriculture and urban expansion.

Military Activities

The large number of military training areas located in the southeastern coastal plain results in significant impacts on old-growth pine forest. Both a reduction in vegetative ground cover and changes in species composition can result from routine operations and military training activities. Concerns for the impacts of tracked vehicle activity and artillery and aircraft noise on the red-cockaded woodpecker recently prompted a Department of Defense conference on the management of this endangered species (Doug Ripley, personal communication).

Mitigations of Impacts

The conservation of habitats requires consideration of mitigations for the major activities impacting habitats of concern. In the *Southeastern Forests and Croplands*, the primary habitat impacts are caused by the following:

- Timber harvesting of old-growth or mature forests.
- Land conversion of scrub, coastal, and wetland habitats.
- Fragmentation of contiguous forest.
- Mining and acidification of Appalachian forest.

Timber Harvesting

At a minimum, the production of commercial wood products from an area must not exceed the sustainable level if the ecological integrity of a forested area is to be maintained. Where sensitive forest types exist, logging may be completely prohibited or constrained to specific methods to prevent habitat loss or degradation. In other areas, more extreme harvesting methods may be allowed or prescribed to establish or maintain desired forest conditions. Acceptable methods will vary according to local forest ecology and the desired future condition of the site. Analysis of harvesting techniques must be based upon an analysis of the structure and diversity of the forest canopy, midstory, and understory.

A recent directive of the Chief of the U.S. Forest Service acknowledges this fact and points out that clear cutting is acceptable only when needed to replicate natural ecological processes. Selective cutting can preserve forest structural diversity, the primary determinant of wildlife habitat (Harris et al. 1979). However, it can reduce horizontal diversity (NRC 1982). The harvesting technique employed must be based upon sound silvicultural prescriptions and demonstrate its capability to maintain vertical diversity (foliage height diversity), horizontal diversity (interspersion, edge, juxtaposition, patchiness), and a mixture of live and dead wood. Specific timber harvesting operations should be designed to preserve the structure and diversity of the natural forest habitat.

An important component of selective cutting should be the preservation of standing dead trees. Many species of birds nest, roost, or forage for invertebrates in standing trees with decayed wood. These cull trees are usually the first focus of forest thinning operations to the detriment of the birds. Breeding bird abundance declines rapidly following a clear cut, and the species composition continues to change for 10 to 15 years (DeGraaf 1991). However, if trees with cavities are saved, many of these species can successfully forage on sound boles. .About one large cavity or den tree per 2 ha is required for populations of large species such as wood ducks; this requires harvest rotations of 100 to 125 years (although rotations of 65 years produce trees large enough for smaller cavity species).

Timber harvesting practices modified to reduce the impacts of simplification must also address fragmentation. As an example, fragmentation has been especially severe in southeastern bottomland forests (Gosselink and Lee 1987). In this case, the setting aside of undisturbed tracts will not suffice to achieve viable populations of the larger, wider-ranging species. Not only do some species require specific habitat conditions (such as forest-interior species like Bachman's warbler), but others require particular arrangements of several communities. Therefore, a successful faunal conservation strategy must emphasize the landscape configuration, not just the structural content of the communities themselves.

Responding to the "biodiversity crisis," the U.S. Forest Service is moving toward an ecosystem approach to forest management (Bob Szaro, personal communication). Recent forest management plans have incorporated tenets of the "New Forestry" espoused by Jerry Franklin. These progressive plans require the rigorous implementation of ecological management practices to maintain forest productivity and preserve the functioning of sensitive forest components such as old-growth or late-successional forests. Effective mitigations for habitat conservation in forest management require specific management measures at the site, watershed, and landscape levels. For example, the location and size of timber harvests should be planned to minimize reduction of core area of mature forest (e.g., harvest only alternate basins until regrowth). Maintenance of mature-forest stands in managed landscape can be achieved by extending rotation (beyond 80) to 150 to 200 years, by leaving some stands unharvested for old growth, and by linking stands. Landscape-scale considerations include the provision of buffer zones and habitat corridors as discussed in the introduction of this document. Management measures recommended for conserving habitat within managed forests include the following:

- Minimize the construction of new roads and close roads not in use either permanently or seasonally.
- Use best management practices (BMPs) such as filter strips to minimize erosion during harvesting or road construction.
- Maintain 100-ft riparian zones with adjacent feathered transition zones to buffer edge effects.
- Restrict harvesting operations to periods when the ground is either dry or frozen.
- Maintain site productivity by retaining large woody material and minimizing mineral soil exposure and compaction during harvesting.
- Manage for natural disturbance patterns to maintain natural openings and successional-stage composition.
- Maintain connections between blocks of interior forest, especially old growth.

- Provide for the protection of special areas, including cliffs, caves, taluses, riparian areas, and old-growth stands.

- Maintain the structural integrity and the native variety of the forest by managing for the natural composition of the following components: vegetative types, seral stages, tree types and sizes, standing dead trees and down material, tree snags, and cavity trees.

Land Conversion

Effective mitigation of land conversion activities can sometimes be obtained only by avoiding impacts on rare or unusual habitat types. Rarely, if ever, is restoration or compensation an adequate mitigation for the loss of these habitats. In these cases, mitigation is a siting issue, where construction and degrading activities are located at a distance from the habitats of concern. The habitat is adequately preserved if all possible impact scenarios are accounted for. Barring this solution, effective management measures must be implemented to ensure the protection of the habitats of concern.

In the case of unique scrub habitats or coastal systems, hydrological and contamination concerns are especially important. Construction or resource management activities require the use of sediment filter strips and other means of intercepting offsite contaminants. Road building and structural "improvements" must not result in altered hydrological regimes. Where rare plant types exist or where habitats are unstable (e.g., bogs and sand dunes), recreational access associated with nearby development may have to be limited.

Amelioration of impacts from land conversion to transportation uses requires special mitigation measures. As with all land conversion, the construction of highways and power-line corridors is primarily a siting issue. Avoidance of sensitive habitats may be accomplished by modifications to the route design, and the extent of disturbance can be limited by careful construction practices. However, fragmentation of the larger area is unavoidable in the case of land conversion to transportation corridors. Many structural mitigation strategies can be used to lessen the impact on animal movement across transportation routes. Primarily, these include the construction of fences and underpasses. The goal of these structural measures should be to mimic the natural movement and migration patterns of the affected species.

Mining

Mitigation of mining impacts involves siting issues, technological solutions to eliminate contamination, and restoration programs. The major mitigations for oil and gas extraction and production are the proper sitings of rigs, reserve pits, processing facilities, and roads where they will have minimal impacts on habitats of concern. Most important for coal and mineral mining is the siting of mining operations and tailing ponds to avoid habitats of concern, wetlands, riparian areas, and recharge areas. Specific mitigation measures depend on the type of mining and the specific process causing impacts. It is generally best to minimize the area affected as it is unlikely that even the disrupted soils and sediments can be restored. In addition to minimizing the area disturbed, activities should be timed to avoid disturbing nearby plants and animals during crucial periods of their life cycle. Possible mitigation measures for mining operations include the following (SAIC 1991a, 1991b):

- Design of mine entrances and workings to minimize future mine drainage.
- Runon and runoff control measures such as berms and ditches.
- Adequate depth and lining of pits for containment of muds and leachate.
- Elimination of fluid migration through casings and dewatering.
- Separation of wastes and contaminated soils with proper disposal.
- Treatment of leach heaps and neutral or acidic wastewaters to reduce the load of cyanide, nitrates, and heavy metals.
- Closure planning that addresses hydrology, geochemical controls, treatment, and restoration.
- Nets or other covers over process ponds.
- Maintenance of an anaerobic environment in the tailing pile during periods of inactivity.
- Secondary containment of tanks and contingency plans for sudden or catastrophic releases.
- Backfilling and sealing of the mine workings during mine reclamation/closure.
- Recycling of process water, smelter slag, and air pollution control dust.
- Monitoring and elimination of discharges to surface water, groundwater, soils, and air.
- Replenishment of surface and ground waters with treated effluents.
- Road closure and reclamation (following recontouring) with revegetation of native species.

Although the reclamation of mined lands is often unsatisfactory for ecological habitat restoration, reforestation with native trees has been demonstrated (Plass 1973) and would serve to reduce the abundance of nest parasitic brown-headed cowbirds and restrict their access to mature forest.

Military Activities

Mitigation of the impacts of military activities on habitats has only recently received attention. The Army Corps of Engineers' Construction Engineering Research Laboratory in Champaign, IL, is developing a Land Condition-Trend Analysis (LCTA) Program (Diersing et al. 1992) as a comprehensive means of matching military training mission objectives with effective natural resource management. If such a plan is instituted, it is likely that careful coordination of the siting and timing of training operations will dramatically reduce habitat impacts. An awareness of the ecological consequences of specific activities is essential to effective mitigation. The following general mitigation measures apply to the primary impacts of military activity:

- Timing and siting of operations - The noise and disturbance associated with aircraft flights and large troop maneuvers cannot be eliminated. However, sensitive environments can be avoided and operations can be timed to avoid critical nesting and migratory periods.

- Calculation of allowable use for tracked vehicles - Tracked vehicle movements are a major cause of habitat degradation. Vegetation destruction and soil erosion and compaction are the primary impacts. Precise equations can be developed that estimate sustained tracked vehicle use based on physical properties of the environment, vegetative cover, and changes in vegetative cover caused by the passage of tracked vehicles. For example, tracked vehicle use should be restricted to all-weather roads when possible.

- Fire suppression during artillery practice - Fires created by artillery pose a major problem in certain environments. Rapid identification and suppression by helicopter can virtually eliminate the spread of large-scale fires.

Guidelines for Reviewers

Reviewers of environmental impact assessments will find this document useful if they follow the steps laid out in the introduction:

1. Review the status and trends of habitats in the region.

2. Identify the habitats of concern.

3. Link the activities involved with impacts to these habitats of concern.

4. Devise appropriate mitigations for the impacts.

Each reviewer can then determine the adequacy of the environmental impact assessment in question and recommend modifications to enhance its effectiveness.

In identifying the habitats of concern, the reviewer should supplement the information in this document with detailed locational information on the abundance and distribution of habitats within the region of interest, and with any historical information on the extent and quality of these habitats. Most important, the reviewer should characterize the habitats in terms of their ecological values (e.g., use of wooded wetlands by migratory waterfowl).

In considering the links between activities and habitats, the reviewer should look beyond direct impacts to indirect and subtle effects, including cumulative impacts, interactive and synergistic impacts, and scale-dependent impacts (e.g., effects of fragmentation on ecosystem integrity and species home ranges).

In devising possible mitigations, the reviewer should follow the seven principles for habitat mitigation repeated below. The reviewer should also determine whether adequate assurances have been given that the mitigations proposed will be completed.

1. Base mitigation goals and objectives on a landscape-scale analysis that considers the needs of the region.

2. Mimic natural processes and promote native species.

3. Protect rare and ecologically important species and communities.

4. Minimize fragmentation of habitat and promote connectivity of natural areas.

5. Maintain structural diversity of habitats and, where appropriate, species diversity to promote the natural variety of the area.

6. Tailor management to site-specific environmental conditions and to the unique impacts of the specific degrading activity.

7. Monitor for habitat impacts and revise mitigation plans as necessary.

Finally, the reviewer should consider the proposed activities and mitigations in the context of relevant regional program goals and objectives (e.g., whether the outcome of the project will be in accordance with principles set out by regional planning commissions such as those established for the Chesapeake Bay and the Gulf of Mexico).

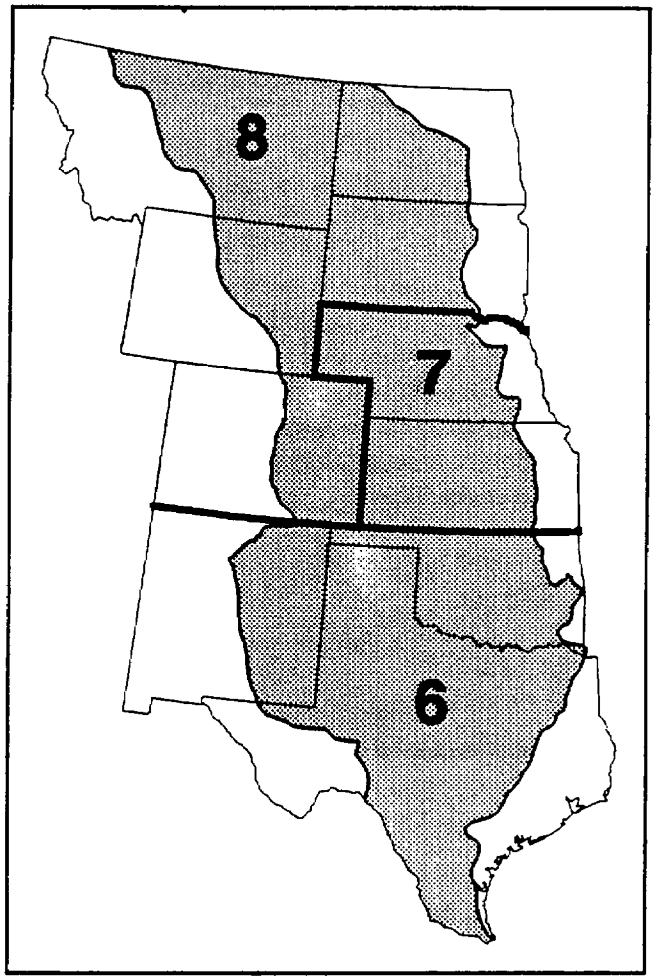
Contacts and Information Sources

When considering habitat conservation issues in an environmental impact assessment for the *Southeastern Forests and Croplands*, the reviewer should consult the following organizations and individuals for information on habitat impacts and mitigations:

State Natural Heritage Programs
U.S. Fish and Wildlife Service, Regional and Area Offices
State Fish and Game Departments
University and Research Programs
Herbaria and Museums

Dorothy Allard, Regional Ecologist, The Nature Conservancy
W.T. Olds, Associate Regional Director, Fish and Wildlife Enhancement, U.S. Fish and Wildlife Service

Great Plains Habitat Region: *Great Plains and Prairies*



Geographical Description of the Region

The Great Plains Habitat Region, *Great Plains and Prairies*, contains parts of 10 states. The region includes parts of North Dakota, South Dakota, Montana, Wyoming, Colorado, Nebraska, Kansas, Oklahoma, New Mexico, and Texas. Parts of EPA Regions 6, 7, and 8 are included. The accompanying map indicates the boundaries of this habitat region and the states it comprises.

The *Great Plains and Prairies* comprises 14 ecoregions (Omernik 1987). The vegetation of the region includes a range of grama, needlegrass, wheatgrass, Nebraska sand hills prairie, bluestem, buffalo grass, indiangrass, bluestem prairie (bluestem, panic, indiangrass), cross timbers (oak, bluestem), mosaic (bluestem, oak, hickory), Blackland prairies of wheatgrass, fescue, sandsage, juniper, oak savanna, mesquite acacia, and savanna bristlegrass. The land use patterns comprise croplands, croplands with grazing lands, cropland with pastures, subhumid grasslands, and semi-arid grazing lands, irrigated agriculture, woodlands, forests, and open woodlands grazed.

Habitats of Concern

The *Great Plains and Prairies* contains many habitats of concern, of which the most obvious fall into four general categories: riparian habitats, prairies, brushland, and wetlands. The principal habitats of concern most at risk in the *Great Plains and Prairies* are listed below.

PRINCIPAL HABITATS OF CONCERN
IN THE *GREAT PLAINS AND PRAIRIES*

1. Riparian habitats
   - hardwood draws

2. Prairies
   - tallgrass prairie remnants in Kansas
   - short and midgrass prairie (North Dakota, South Dakota, Colorado, Nebraska, Kansas)
   - Texas blackland prairie and other types

3. South Texas brushlands

4. Wetlands
   - prairie potholes (Montana, North Dakota, South Dakota)

Habitat Values and Trends

The term "rangeland" describes the lands with climate or soil conditions unsuitable for tree growth. Rangeland comprises nearly a billion ac (34% of land area) in the United States, including some of the world's most productive rangeland (Box 1989).

Grasslands

The *Great Plains and Prairies* contain 78 million ac of rangeland (USDA Forest Service 1989), including both the true prairie (tallgrass) and plains grassland (shortgrass). Tallgrass prairie is dominated by bluestem grasses and includes prairie potholes important for waterfowl breeding. Most of the original tallgrass prairie was plowed under, and the remaining areas were invaded by trees following fire suppression. The largest existing area of tallgrass prairie (1.5 million ha) covers the Flints Hills of Kansas and the Osage Hill of Oklahoma. Plains grassland is dominated by short warm-season grasses of blue grama and buffalo grass and supports pronghorn, mule deer, white-tailed deer, jackrabbit, prairie dog, greater prairie chicken, and sharptailed grouse. The decline of the long-billed curlew is associated with the decrease in this habitat.

About 84% of mammal species and 74% of avian species are associated with rangeland ecosystems during at least part of the year. Thirty-eight percent of the nation's fish species and 58% of the amphibians are represented in the relatively arid rangeland ecosystems (Flather and Hoekstra 1979).

Based on 1984 maps, the average number of endangered and threatened species per county is 3.3 for the *Great Plains and Prairies*. Perhaps the most important habitat for animals in the *Great Plains and Prairies* are riparian areas where the juxtaposition of terrestrial and wetland or aquatic systems enhances the value of the habitat.

By the beginning of the 20th century, the American range was generally overgrazed and depleted. Severe droughts also contributed to the deterioration of rangeland. Although the total area of rangeland has remained relatively constant, the condition of the range ecosystems has varied considerably with competition by livestock for forage and other factors. Cattle, sheep, and wild horses and burros have contributed to reduced forage and to changes in vegetation composition on the majority of U.S. rangelands. Many native prairie types have been lost to overgrazing or agricultural conversion. The loss of grassland habitat has been responsible for declines in many bird populations. The mixed prairie or shortgrass prairie is subject to drought, grasshoppers and jackrabbit attacks, and cacti invasion. However, native shortgrasses are outstanding in their resistance to grazing (perhaps developed in response to grazing by bison) and have shown remarkable improvement in certain areas. An increase in rangeland area in the Great Plains of 11 million ac is predicted for the next 50 years as a result of the natural succession of agricultural land in the Conservation Reserve Program (Joyce 1989). Rangeland in Texas and Oklahoma will likely increase by 14 million ac or 11% during this period.

Texas Habitats

Within the *Great Plains and Prairies*, Texas contains a greater variety of habitats than any other state. However, virtually all of the blackland and tallgrass prairie, coastal bottomlands, and low hills in Texas have been converted to farms, cities, and suburbs (Loftis 1991). Less than 1% of blackland prairie remains in north-central Texas. In the Lower Rio Grande Valley, there remains less than 2% of the native scrubby, hot delta that once was nearly as rich in wildlife as the Everglades. In particular, duck populations have declined, bird variety in the valley has decreased, and the ancient gene pools of blackland prairie plants are being lost. Brushlands in south Texas still support endangered cats (jaguarundi and ocelot) and numerous subtropical bird species. Past brush clearing activities have greatly impacted this habitat, although the U.S. Fish and Wildlife Service is currently preserving and restoring brush habitat in the Lower Rio Grand Valley.

Within Texas the greatest loss of natural vegetation has occurred in the state's High Plains and Blackland Prairies regions. The following describes the status of the natural regions of Texas within the *Great Plains and Prairies* (Loftis 1991):

- High Plains - Lost the buffalo and pronghorn with conversion to cattle and crops. Damming of rivers has eliminated the willow and cottonwood and replaced them with the Old World exotics, salt cedar, and Russian olive.

- Rolling Plains - Low hills and broad flats with headwaters of major rivers. Native grasses have been cleared and replaced with mesquite, snakeweed, and prickly pear.

- Edwards Plateau - Limestone hills, springs, and rivers support endangered wildlife; ranches and big cities compete with wildlife for ground water.

- Cross-Timbers and Prairies - Strips of prairie crossed by oak forests have been changed by farming and urban development.

- Blackland Prairies - Originally 12 million ac, the tallgrasses, big bluestem, Indiangrass, little bluestem and gammagrass are near extinction at 5,000 ac.

- Post Oak Savannah - Nearly all of original grasslands have been plowed under or invaded by thickets.

- Rio Grande Plain - Open grasslands have been converted to thorn forest by overgrazing, and less than 1% of the natural habitat remains.

Riparian Areas

Riparian areas in the *Great Plains and Prairies* constitute perhaps the region's most important habitat type. Although they represent only 2% to 4% of the land area in the United States, they make up 80% of the wildlife habitat. It has been demonstrated that most endangered species require riparian areas (Johnson 1989). Many neotropical migrants also rely on western riparian areas as critical nesting sites. The value of riparian habitat extends at least 0.25 miles into adjacent areas and can support a density of pairs of breeding birds up to 1,000 per 100 ac (Carothers and Johnson 1975).

Riparian areas provide habitat for more species of birds than all other western rangeland vegetation types combined (Chaney et al. 1990). Although riparian areas cover less than 1% of the West, they also serve important ecosystem functions (Gillis 1991). They keep watersheds healthy by storing and releasing water from spring runoff of snowmelt and summer storms, and by providing watering holes for wildlife as well as cattle. They filter sediment and aid floodplain development, improve floodwater retention and groundwater recharge, develop plant root masses that stabilize streambanks, develop channel characteristics that provide appropriate habitat for fish, and support greater biodiversity.

The linear nature of riparian areas contributes to their value (Gregory et al. 1991). River valleys connect montane headwaters with lowland habitats, and provide for the transfer of water, nutrients, sediment, particulate organic matter, and organisms. Riparian areas transfer these materials laterally onto floodplains and create complex mosaics of landforms and heterogeneous ecosystems. Wildlife utilize riparian areas for food, cover, nesting, and rearing of young. Riparian habitats are frequently used by wildlife as migration routes (Thomas et al. 1978). The greater heterogeneity of vegetation in unaltered riparian habitat increases the available ecological niches and increases the number of species that can be supported.

Johnson (1978) estimates that only 10% of the original riparian habitat in United States remains, and that 6% is lost annually. In the Great Plains, less than 1% of land is riparian vegetation (Crouch 1978). Major losses resulted from drainage for conversion to agriculture; other causes include channelization for navigation and flood control, flooding caused by dam construction, and diversion of streamflow for irrigation. Alterations include grazing, timber harvesting, road construction, mining, and other impacts.

Wetlands

Specific national wetland problèm areas identified by Tiner (1984) in the *Great Plains and Prairies* include the following:

- The emergent wetlands of the Prairie Pothole Region.
- Wetlands of the Nebraska Sandhills and Rainwater Basin.

The drainage of inland marshes for range and agriculture has been the greatest in the prairies of the Dakotas and Minnesota, the sandhills of Nebraska, and the Florida Everglades. In Texas, wetlands covering 8.4 million ac or 52% of the original extent have been lost. One-third loss of this loss (296,132 ac) has been in the playa lake complexes that are especially important for waterfowl and migratory species. In general, emergent wetlands have high priority in this region owing to their functional importance and the constant threat of degradation.

Activities and Impacts Affecting Habitats

IMPACTS ON HABITATS OF CONCERN IN THE *GREAT PLAINS AND PRAIRIES*				
	Land Conversion	Timber Harvesting	Grazing	Water Management
Riparian habitats	Residential development and construction of pipeline and transportation corridors	Minor	Sever overgrazing and physical habitat degradation	Impacts of damming and water diversions
Prairies	Conversion to agriculture	None	Severe overgrazing	Moderate
Brushlands	Conversion to urban uses	None	Minor	Minor
Wetlands	Conversion to agriculture and urban uses	None	Minor	Major

The *Great Plains and Prairies* rangeland areas are at risk principally from grazing and water management projects. Dam construction in the Platte River area has also been a major source of modification to terrestrial habitat in that area. Of special concern are the remnants of the tallgrass prairie ecosystem, which has suffered extensive conversion. The rarest of all North America's major biomes, only 10% of the original 142 million ac of tallgrass remains. Much of the 10% represents fragments of old railway rights-of-way, pioneer cemeteries, and various preserves. This prairie habitat is at risk from human encroachment and cattle grazing.

This region is experiencing rapid population growth as part of the westward migration. Highway construction, in particular, has expanded and is creating substantial cumulative impacts on natural areas. The Texas hill country is being rapidly converted to urban uses. Riparian areas are being degraded through overgrazing, and prairie potholes are being converted to agriculture. Although the region has a relatively small population, urban areas such as Denver, CO, and central Texas are experiencing rapid growth while second-home and time-share development is occurring in previously pristine areas (e.g., Montana, Flathead Mountains in Wyoming, and Colorado prairie river systems).

Grazing and water projects especially threaten riparian environments throughout the region. For example, overgrazing and phreatophyte control are destroying riparian vegetation. Water diversions have caused major losses of riparian and wetland habitats and are contributing to the declines of waterfowl along the Mississippi Flyway.

<u>Land Conversion</u>

To date, the most fertile soils within the *Great Plains and Prairies* have been converted to croplands; these same areas have historically supported the greatest abundance of wildlife (Mayer and Laudenslayer 1988). In addition, urban development has been a major source of rangeland conversions. Pressure on local governments to convert open space to residential, commercial, and industrial uses to accommodate growth has been intense, and will continue to destroy rangeland habitats where population growth is most pronounced.

Conversion of rangelands to cropland will increase with the availability of ground water for irrigation (USDA Forest Service 1989). In particular, sandy rangeland in Texas, Colorado, New Mexico, and Nebraska has been converted to farmland (Sheridan 1981). Abandonment of these farms can lead to desertification if the ground water has been depleted. Areas of concern for desertification include Kiowa and Crowley Counties in Colorado. In these semiarid lands, land conversion to agriculture, grazing, and water management can cause groundwater overdraft, salinization of topsoil and water, reduction of surface water, high soil erosion, and destruction of native vegetation.

As with forest habitats, the spatial pattern and fragmentation of rangeland vegetation can negatively affect native fauna and ecosystem health. The loss of grassland habitat to agriculture is responsible for the decline in prairie birds, especially those requiring large continuous habitats, and is analogous to the reduction in old-growth forests and the decline in its obligate species. The upland sandpiper, bobolink, dickcissel, grasshopper sparrow, savannah sparrow, and Henslow's sparrow all declined by 90% between the 1950s and 1970s (Graber and Graber 1983).

<u>Agricultural Impacts</u>

The intensive use of agricultural land in certain areas of the *Great Plains and Prairies* pose additional stresses to habitats through cultivation and irrigation practices (NRC 1982). The use of fertilizers and pesticides, irrigation and drainage, double cropping and increased field size all contribute to increased pollutant loads and severe impacts on habitats. Agricultural chemicals are toxic to many species and can negatively affect population levels, community composition, and ecosystem dynamics. Other intensive cultivation practices directly reduce important hedgerow and riparian habitat and usually produce severe offsite impacts.

Grazing

Widespread devastation of rangeland resulted from uncontrolled overgrazing between 1880 and 1935, and the damage was amplified by the drought years of the 1930s (Branson 1985). The enactment of the Taylor Grazing Act of 1934 reduced grazing pressure at that time. With the advancement of range management science and the moist years following 1960, considerable improvement occurred in range vegetation. However, the USDA Forest Service (1989) reports that 21% of its rangelands are still in "unsatisfactory" condition. The Bureau of Land Management (1989) reports that BLM rangeland condition is 33% good or better, 38% fair, and 13% poor.

The management of public land grazing is shared between the land management agency and the grazing permittee. Grazing permits are issued, and allotments are inspected for use, condition, and compliance by the management agency; actual management of the livestock and maintenance of improvements is the responsibility of the permittee. Attempts to reduce grazing allotments in national forests to allow improvements on lands in poor or fair condition has caused resentment among graziers. However, federal permit fees are only one-fifth the rate for private lands. As private grasslands continue to decline in acreage as a result of urban and agricultural conversion, there will be increased pressure on public lands.

Grazing poses the following threats to rangeland habitats (Cooperrider 1990):

- Competition with ungulates and small herbivores (e.g., desert tortoise) and limits on the populations of free-roaming pronghorn antelope, mule deer, elk, and bighorn sheep.
- Transmission of disease (e.g., dramatic diebacks in bighorn sheep with domestic sheep grazing).
- Loss of cover for birds.
- Spread of exotics and noxious weeds.
- Desertification, or serious degradation.
- The conversion of lands with sagebrush and pinyon-juniper to reseeded grassland for more forage.

Riparian Areas

The most severe impact in terms of supporting healthy ecosystems and native faunas on rangelands has been the loss of 70% to 90% of riparian areas to human activities (Ohmart and Anderson 1986). Losses of riparian areas have caused the endangerment of habitat-dependent species and likely will cause the extirpation of many species if the last remaining areas of individual habitat types are lost (e.g., 10 species may go extinct if the cottonwood-willow association disappears). Johnson (1978) estimates that 6% of riparian areas continue to be lost annually through water management activity, grazing, sand and gravel extraction, and development activities.

On average, the riparian zone is only 2% of a grazing allotment, but it produces 20% of the forage, and the cattle consume 80% of their forage from these riparian areas. Stream bottoms are natural concentration areas for livestock seeking succulent forage, shade, reliable water supply, and favorable microclimate. Only when access is limited by steep slopes are livestock absent from unfenced riparian areas. Grazing impacts riparian areas both by removing vegetation and by trampling. By affecting the spacing of plants, width of the riparian corridor, seedling establishment, and species composition, floristic diversity is often lower in grazed areas. Trampling increases soil compaction, erodes streambanks, decreases water quality, widens and shallows channels, and physically destroys vegetation (Kauffman and Krueger 1984). Riparian degradation causes accelerated runoff and erosion of downcut streambeds, lowered water tables, and desertification of the land. It has a negative impact on wildlife habitats and leads to declines in willows and native grasses. In addition, degraded riparian areas are more susceptible to upland inputs as healthy riparian areas can filter out upland degradation. While the condition of all rangelands has improved since 1980, riparian areas are in their worst historical condition.

Although the values and functions of riparian areas have been widely and severely impacted by cultivation, road building, mining, urbanization, logging, and damming of rivers, grazing has caused the most geographically extensive impacts (Chaney et al. 1990). Impacts of grazing on riparian areas include the following:

- Little vegetation to stabilize streambank and shade stream.
- Lowered water table and subsurface water storage.
- Reduced or absent summer flow.
- Warm water in summer and icing in winter.
- Poor habitat for fish and aquatics.
- Poor habitat for wildlife.
- Reduced amount and quality of forage.

Water Management

The regulation and damming of streams are often performed to control flooding and drain land, resulting in the impoverishment of riparian vegetation (Szaro 1991). Dams and water diversion significantly change downstream flow regimes, levels of winter floodwater, dry-season flow rates, and riparian-zone soil moisture. Downstream areas lose pulse-stimulated responses, while upstream areas are affected by water impoundment and salt accumulation. Native riparian plants are usually unable to colonize the shore of reservoirs because of the altered hydrologic regime. For example, high water levels are maintained much longer in reservoirs than in rivers and streams; changes in the level are more drastic; and the large winter/spring floods required for alluvial seedbeds (e.g., cottonwood) are eliminated.

Mitigations of Impacts

The conservation of habitats requires consideration of mitigations for the major activities impacting habitats of concern. In the *Great Plains and Prairies*, the primary habitat impacts are caused by the following:

- Land conversion of riparian and wetland habitats.
- Grazing of riparian areas.
- Water management impacts of diversion and damming on riparian and wetland areas.

It is likely that certain areas will see additional conversions to cropland or pasture, and that more open ranges will be fenced and thus restrict winter grazing by native ungulates. Increased irrigation will likely follow higher demand for water and adversely affect water tables and stream flow on rangelands. These and other activities will pose a complex of interrelated effects on habitats of concern and will require a holistic, ecosystem-level approach to mitigation. The effects of future management and mitigations on riparian areas will have the greatest impact on wildlife and native ecosystem health (NRC 1982).

Land Conversion

Effective mitigation of land conversion activities can sometimes be obtained only by avoiding impacts on rare or unusual habitat types. Rarely, if ever, is restoration or compensation an adequate mitigation for the loss of these habitats. In these cases, mitigation is a siting issue, where construction and degrading activities are located a distance from the habitats of concern. The habitat is adequately preserved if all possible impact scenarios are accounted for. Barring this solution, effective management measures must be implemented to ensure the protection of the habitats of concern.

In the case of unique riparian or wetland habitats, hydrological and contamination concerns are especially important. Construction or resource management activities require the use of sediment filter strips and other means of intercepting offsite contaminants. Road building and structural "improvements" must not result in altered hydrological regimes. Where rare plant types exist or where habitats are unstable, recreational access may have to be limited. These mitigations can be best implemented by creation of a regional land-use plan (through a coordinating council like the Waterfowl Flyway Council) and landowner incentives (like the Conservation Reserve Program).

Conversion to agricultural land is a special concern in rangelands with increasing irrigation potential. Land conversion to agriculture can cause groundwater overdraft, salinization of topsoil and water, reduction of surface water, high soil erosion, and destruction of native vegetation. Mitigations include more conservative irrigation techniques and improved drainage systems. Soil conservation techniques vary from windbreaks to contour plowing, stripcropping, rotation of crops, conversion to grass, and/or minimum tillage.

Grazing

Future management of grazing on rangelands will determine whether range conditions worsen or improve from their currently degraded state (NRC 1982). In the past, range condition has been estimated by (1) forage production relative to a mythical average, and (2) production of livestock. Recently, some range managers have begun to base condition estimates on deviation from an ideal range or ecological climax. These and other improvements in range science provide for consideration of objectives beyond livestock production. For example, the widely used model of E.J. Dyksterhuis (1949) is based on reversible and gradual community change and is now viewed as inaccurate, as it does not incorporate threshold community shifts (Jahn 1991). The problem for habitat conservation is that the proportion of rangeland climax habitats has greatly decreased, similar to the case with old-growth forest. Although there remain disagreements over proper management methods, it is anticipated that more effective use of ecological analyses of range condition will improve the management of rangelands.

Specific methods of mitigating grazing impacts on rangelands include the following (Branson 1985):

- Proper intensity and season of grazing.
- Practices that improve livestock distribution.
- Control of undesirable species using fire or other appropriate methods.
- Land-surface modification to retain soil moisture for forage production.
- Ecologically based management plans for each site using adequate field data.

Proper grazing management can restore the long-term productivity of most rangelands, but obstacles are grazing tradition, the geographical extent of problem, and the difference between short-term costs and long-term benefits. Successful management requires that traditional intensive measures to increase forage be replaced by different management practices. For example, rest-rotation grazing can improve range conditions, while intensified chemical use and mechanical brush removal to improve forage will likely further degrade range habitats. Certainly, successful rangeland mitigation requires time, flexibility, commitment by graziers, and monitoring and evaluation.

Improvements in the condition of riparian areas will provide the greatest proportional benefit to rangeland integrity and functioning. Szaro (1991) argues strongly for an overall ecosystem approach to research and management of riparian areas. This includes the use of reference sites, a watershed (ecosystem) scale approach, and long time scale considerations (greater than 5 years). Mitigation must consider the following factors:

- Riparian floristic (plant species) diversity should take precedence over structural diversity (vegetation layers and patches) as descriptors of the habitat.

- Wildlife species depend both on floristic composition and on the relationship of riparian areas to animal movement patterns and migratory pathways.

- The distribution of riparian vegetative communities varies with topography and depends principally on elevation.

- Flooding and other natural disturbances are important to riparian systems. They contribute to their status as distinct and highly integrated pockets within other communities.

Successful riparian management requires unique solutions to the specific condition at each site (Chaney et al. 1990). However, general principles include the following:

- Include riparian areas in separate pastures with separate objectives and strategies.
- Fence or herd stock out of riparian areas to let vegetation recover.
- Control the timing of grazing (1) to keep the stock off streambanks that are most vulnerable to erosion, and (2) to coincide with the physiological needs of plants.
- Provide more rest to the grazing cycle to increase plant vigor or to encourage more desirable species.
- Limit grazing intensity.
- Change from cattle to sheep to get better animal distribution through herding.
- Permanently exclude livestock from high-risk and poor recovery areas.

Wetlands

Mitigation of wetlands destruction and degradation is the subject of a growing literature (Kusler and Kentula 1989). Restoration and mitigation banking concepts are still being evaluated as effective mitigation measures for direct wetlands alterations.

Guidelines for Reviewers

Reviewers of environmental impact assessments will find this document useful if they follow the steps laid out in the introduction:

1. Review the status and trends of habitats in the region.
2. Identify the habitats of concern.
3. Link the activities involved with impacts to these habitats of concern.
4. Devise appropriate mitigations for the impacts.

Each reviewer can then determine the adequacy of the environmental impact assessment in question and recommend modifications to enhance its effectiveness.

In identifying the habitats of concern, the reviewer should supplement the information in this document with detailed locational information on the abundance and distribution of habitats within the region of interest, and with any historical information on the extent and quality of these habitats. Most important, the reviewer should characterize the habitats in terms of their ecological values (e.g., use of wooded wetlands by migratory waterfowl).

In considering the links between activities and habitats, the reviewer should look beyond direct impacts to indirect and subtle effects, including cumulative impacts, interactive and synergistic impacts, and scale-dependent impacts (e.g., effects of fragmentation on ecosystem integrity and species home ranges).

In devising possible mitigations, the reviewer should follow the seven principles for habitat mitigation repeated below. The reviewer should also determine whether adequate assurances have been given that the mitigations proposed will be completed.

1. Base mitigation goals and objectives on a landscape-scale analysis that considers the needs of the region.

2. Mimic natural processes and promote native species.

3. Protect rare and ecologically important species and communities.

4. Minimize fragmentation of habitat and promote connectivity of natural areas.

5. Maintain structural diversity of habitats and, where appropriate, species diversity to promote the natural variety of the area.

6. Tailor management to site-specific environmental conditions and to the unique impacts of the specific degrading activity.

7. Monitor for habitat impacts and revise mitigation plans as necessary.

Finally, the reviewer should consider the proposed activities and mitigations in the context of relevant regional program goals and objectives (e.g., whether the outcome of the project will be in accordance with principles set out by regional planning commissions).

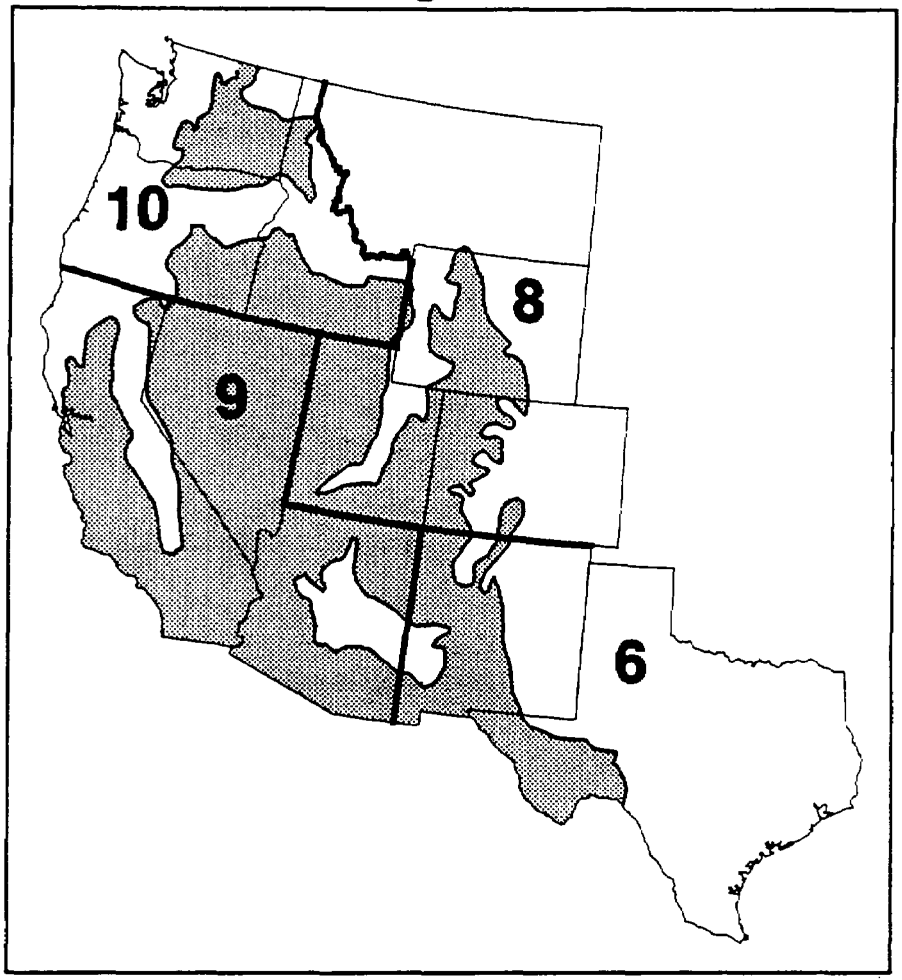
Contacts and Information Sources

When considering habitat conservation issues in an environmental impact assessment for the *Great Plains and Prairies*, the reviewer should consult the following organizations and individuals for information on habitat impacts and mitigations:

State Natural Heritage Programs
U.S. Fish and Wildlife Service, Regional and Area Offices
State Fish and Game Departments
University and Research Programs
Herbaria and Museums

Patrick Bourgeron, Regional Ecologist, The Nature Conservancy
Robert Jacobsen, Regional Associate Director, Fish and Wildlife Enhancement, U.S. Fish and Wildlife Service

Western Rangelands Habitat Region: ***Western Deserts and Grasslands***



Geographical Description of the Region

The Western Rangelands Habitat Region, *Western Deserts and Grasslands*, contains parts of 12 states. The region includes parts of Texas, New Mexico, Arizona, California, Nevada, Utah, Colorado, Wyoming, Idaho, Montana, Oregon, and Washington. Parts of EPA Regions 6, 8, 9, and 10 are included. The accompanying map indicates the boundaries of this habitat region and the states it comprises.

The *Western Deserts and Grasslands* comprises 11 ecoregions (Omernik, 1987). The natural vegetation included in the Region consists of a variety of sagebrush steppe (sagebrush and wheatgrass), saltbush, greasewood, creosote bush, bur sage, needlegrass shrub steppe, juniper, pinyon woodlands, blackbush, Great Basin sagebrush, grama, tobosa shrub steppe, Trans-Pecos shrub savanna (tarbush, creosote), chaparral (manzanita, ceanothus, chamise), and tule marshes (bulrush and cattails). The land use pattern is mostly desert shrublands both grazed and ungrazed, irrigated agriculture, open woodlands grazed, subhumid grasslands, semi-arid grazing lands, forests and woodlands mostly ungrazed, and croplands with grazing land.

Habitats of Concern

The *Western Deserts and Grasslands* contains many habitats of concern, of which the most obvious fall into five general categories: riparian habitats, wetlands, desert complexes and scrub habitats, grasslands, and forested habitats. The principal habitats of concern most at risk in the *Western Deserts and Grasslands* are listed below.

PRINCIPAL HABITATS OF CONCERN
IN THE *WESTERN DESERTS AND GRASSLANDS*

1. Riparian habitats

2. Wetlands
   - shallow emergent wetlands in pluvial lake basins
   - cold desert and steppe in Great Basin and Intermountain regions
   - saline desert wetlands

3. Desert complexes and scrub habitats
   - Sonoran and Mojave desert communities
   - sagebrush type
   - salt desert shrub type

4. Grasslands
   - mixed prairie or shortgrass prairie
   - California grassland
   - Palouse grassland of the Northwest
   - southwest semidesert grassland

5. Forested habitats
   - pinyon-juniper woodland
   - ponderosa pine-bunchgrass type
   - chaparral vegetation of Arizona and California

Habitat Values and Trends

The term "rangeland" describes the lands with climate or soil conditions unsuitable for tree growth. Rangelands encompass nearly a billion ac (34% of land area) in the United States, including some of the world's most productive rangeland (Box 1989). *Western Deserts and Grasslands* habitats traverse the entire range of life zones from the alpine communities of high mountains to the subtropical Sonoran Desert scrub plains and valley of the lower Gila and Colorado Rivers. In the Rocky Mountain region, rangelands (including pinyon-juniper and chaparral-mountain scrub forests) comprise about 336 million ac. Sagebrush alone constitutes the second largest habitat type in United States with 105 million ac, while other habitats include southwestern shrubsteppe, desert shrub, mountain grasslands, mountain meadows, desert grasslands, and plains grasslands. Rangelands in the Pacific States total 68 million ac with 23 million ac in grassland and 45 million ac in shrubland (USDA Forest Service 1989).

By the beginning of the 20th century the American range was generally overgrazed and depleted. Severe droughts also contributed to the deterioration of rangeland. The majority of rangeland is in the West, where declines in area have been minor--4% in the Rocky Mountains and 5% in the Pacific States (USDA Forest Service 1989). Although the total area of rangeland has remained relatively constant, the condition of the range ecosystems has varied considerably with competition by livestock for forage and other factors. Cattle, sheep, and wild horses and burros have contributed to reduced forage and to changes in vegetation composition on the majority of U.S. rangelands. Many native prairie types have been lost to overgrazing or agricultural conversion. Grazing and fire suppression have allowed brush species to replace many of the grass forage species on 200 million ac of the Southwest (National Association of Conservation Districts 1979), negatively impacting bighorn sheep, pronghorn, sage grouse, masked bobwhite quail, and northern aplomado falcon. At the same time, range management activities (such as pinyon-juniper removal, exotic species plantings, predator and native ungulate control) and development along valleys and lower slopes have affected wildlife community composition and critical winter range for wild ungulates. The loss of grassland habitat has been responsible for declines in many bird populations.

No data exist on the extent of areal changes, but the range of pinyon-juniper has certainly increased since settlement as a result of overgrazing, fire suppression, and climate changes. Projections for the next 50 years indicate that rangeland area will increase by 7 million ac in the Rocky Mountains and 3 million ac in the Pacific States as a result of conversion of agricultural lands through the Conservation Reserve Program (USDA Forest Service 1989). However, even where there have been increases in total area, the condition of these rangelands has been severely degraded. The majority of rangeland on nonfederal and Bureau of Land Management lands is in fair to poor condition (Joyce 1989). In the 11 western states, range conditions on public lands are rated as 2% excellent, 29% good, 42% fair, and 26% poor (Wald and Alberswerth 1989).

Klopatek et al. (1979) demonstrated that the tule marsh ecosystem in California, Nevada, and Utah has suffered the greatest loss of any habitat since presettlement times (89%), primarily owing to agricultural conversion. However, in general, vegetation in the western United States has exhibited the least losses due to land conversion and suffer primarily from degradation. Alpine meadows and barrens have undergone the least change because of their rugged topographical setting. In contrast, riparian areas are especially important to wildlife, and losses of this type of vegetation to human activities are estimated at 70% to 90% (Swift and Barclay 1980). In Texas, important rangelands include the rocky landscape along the Big Bend in the Trans Pecos Region and the extremely diverse Mountains and Basins Region, where overgrazing has damaged most of the desert grasslands and small streams (Loftis 1991).

About 84% of mammal species and 74% of avian species are associated with rangeland ecosystems during at least part of the year, and 38% of the nation's fish species and 58% of the amphibians are represented in the relatively arid rangeland ecosystems (Flather and Hoekstra 1989). Based on 1984 maps, the average number of endangered and threatened species per county is 6.1 for the *Western Deserts and Grasslands*, the highest in the nation. Although most of the value placed on rangeland habitats centers on the grass and shrub vegetation existing under different climatic conditions, and the grazing fauna they support, many other values such as reclusive reptile species and the characteristic cryptogamic crusts of the desert are being recognized. Perhaps most important are riparian areas where the juxtaposition of terrestrial and wetland or aquatic systems enhances the value of the habitat.

Woodland and Shrubland

Pinyon-juniper woodland is a widely distributed vegetation type that supports mule deer, mountain lion, coyote, bobcat, jackrabbit, and numerous birds. Pinyon-woodland has invaded grassland areas owing to lack of fire, seed spread by livestock, overgrazing and reduced competition from grasses, and shifts in climate (Branson 1985). Woodland invasion of big sagebrush has occurred more slowly, usually where pinyon-juniper is often adjacent to sagebrush on the dissections of western basins and mountains. Fire management in now being used to encourage the reestablishment of natural vegetation and native diversity in these areas.

In Arizona and California, chaparral vegetation consists of dense stands of evergreen shrubby vegetation. In California, the sparse herbaceous understory of chaparral is less affected by livestock grazing than grasslands, but alien herbaceous species have largely replaced native perennials in both systems (Branson 1985). Areas in Arizona with high grass were converted to dense chaparral with intensive grazing following mineral prospecting in 1890; other chaparral in the Sierra Nevada is a subclimax of forest maintained by frequent fires. This habitat provides watershed protection and critical habitat for the California condor.

Grasslands

Mountain grasslands provide critical winter range for big game. These mountain meadows are sensitive to abuse, as some are destroyed by roads and camping as well as grazing. Desert grasslands consist of blue and black grama grasses and invading shrubs resulting from increased livestock grazing, climatic change, increased competition among plant species, rabbits and rodents, and fire control. They support pronghorn and collared peccary.

The mixed prairie or shortgrass prairie is subject to drought, grasshopper and jackrabbit attack, and cacti invasion. Native shortgrasses are outstanding in their resistance to grazing (perhaps developed in response to grazing by bison) and have shown remarkable improvement in certain areas.

Nowhere else in the West has the native vegetation been as completely replaced as in the 30-million-ac extent of grasslands in California (Branson 1985). Native perennials were largely replaced by introduced Mediterranean annuals by the 1860s, so that now less than 5% of the current species are perennials. This has been attributed to past overgrazing or perhaps fire. Most of the open grassland in the Sacramento and San Joaquin Valleys is now cultivated or in urban or industrial use. Adjacent grass-woodland and chaparral are grazed by livestock.

The Palouse grassland of the Northwest is dominated by bluebunch wheatgrass on 12 million ha of the Columbia Basin Province of Oregon, Washington, and Montana. Because few ungulates were present before the introduction of domestic stock, native grass species were not resistent to grazing and were strongly impacted by livestock grazing and the invasion of Mediterranean annuals (Branson 1985). The most fertile areas have been cultivated, including some drier lands now irrigated. Grazing is now much reduced in the Palouse grassland, and some improvement in range conditions has occurred.

The widespread change of southwestern semidesert grassland to shrubland is one of the greatest modifications of vegetation on western rangelands. Cited causes include excessive use by domestic stock and the reduction of range fires; the loss of topsoil may prevent ever restoring the original grasslands (Branson 1985). Over the last 100 years, mesquite, creosote bush, and tarbush have expanded to cover the entire range.

Deserts

Four major deserts occur in the western United States: the Sonoran, Mojave, Chihuahuan, and Great Basin Deserts. Among desert habitats, the desert riparian and palm oasis habitats support the greatest number and densities of bird species (Mayer and Laudenslayer 1988). The Sonoran and Mojave Deserts, in particular, support unusual plant and animal communities that are threatened by increased human activities in these regions. Cold desert types of the Great Basin support mule deer, pronghorn, coyote, collared peccary, and feral horses. Hot desert shrublands support desert mule deer, collared peccary, antelope, and desert bighorn sheep.

Both decreased rainfall in this century and effects of grazing have impacted the widely spaced woody plants and cacti of the Sonoran Desert, including the cessation of reproduction in saguaro cactus. The Mojave Desert is suffering degradation from offroad vehicles, which resulted in the cessation of the annual Barstow to Vegas motorcross (The Washington Post 1990). Desert habitats in general support many populations of unique and endangered species, including the desert tortoise. Unique geomorphological features such as desert buttes and the Utah salt flats are also facing threats from recreational activity, air pollution, and water withdrawal (Lancaster 1991).

The sagebrush habitat type is unusually susceptible to change when grazed. Many bunchgrasses in the sagebrush type lack resistance, and the historical response has been the following: (1) an increase in native shrubs undesirable for browsing, (2) reduction in grasses and forbs, and (3) exploitation of voids by alien annual weeds adapted to heavy grazing. A history of grazing and cultivation has led to encroachment and takeover by annual grasses, primarily cheatgrass. Mitigation includes burning of annuals but is effective only where there is sufficient annual precipitation. The success of cheatgrass has facilitated the successful introduction of exotic chukar partridge and supports the majority of wild horse and burro herds. The sagebrush types also support sage grouse, pronghorn, and mule deer. It is likely that the original sagebrush habitat can never be restored to pristine conditions even with removal of domestic animals (Branson 1985).

The salt desert shrub type is often called the shadescale zone because of its sparse vegetation and usually widely spaced shrubs with essentially no understory or interstitial species. In general, where there is an understory (such as black sage), historical overgrazing has reduced grasses and promoted shrub growth and invasion by the exotics halogeton and Russian thistle.

Riparian Areas

Riparian areas in the West constitute perhaps the region's most important habitat type. Although they represent only 2% to 4% of the land area in the United States, they make up 80% of the wildlife habitat. It has been demonstrated that most endangered species require riparian areas (Johnson 1989). Many neotropical migrants also rely on western riparian areas as critical nesting sites. The value of riparian habitat extends at least 0.25 miles into adjacent areas and can support a density of pairs of breeding birds up to 1,000 per 100 ac (Carothers and Johnson 1975).

Riparian areas provide habitat for more species of birds than all other western rangeland vegetation types combined (Chaney et al. 1990). Within the Great Basin of southeastern Oregon and in southeastern Wyoming, more than 75% of terrestrial wildlife species depend on riparian systems. In Arizona and New Mexico, 80% of all vertebrates use them for at least half of their life cycle and more than 40% of the species are totally dependent on riparian areas. Although riparian areas cover less than 1% of the West, they also serve important ecosystem functions (Gillis 1991). They keep watersheds healthy by storing and releasing water from spring runoff of snowmelt and summer storms and by providing watering holes for wildlife as well as cattle. They filter sediment and aid floodplain development, improve floodwater retention and groundwater recharge, develop plant root masses that stabilize streambanks, develop channel characteristics that provide appropriate habitat for fish, and support greater biodiversity.

The linear nature of riparian areas contributes to their value (Gregory et al. 1991). River valleys connect montane headwaters with lowland habitats, and provide for the transfer of water, nutrients, sediment, particulate organic matter, and organisms. Riparian areas transfer these materials laterally onto floodplains and create complex mosaics of landforms and heterogeneous ecosystems. Wildlife utilize riparian areas for food, cover, nesting, and rearing of young. Riparian habitats are frequently used by wildlife as migration routes (Thomas et al. 1978). The greater heterogeneity of vegetation in unaltered riparian habitat increases the available ecological niches and increases the number of species that can be supported.

Of the 175 million ac of floodplains along streams and rivers in the conterminous United States, 20% are considered to be rangeland (Johnson 1978). Valley trenching starting in the 1880s resulted in the loss of many riparian meadows through massive sheet and rill erosion. The introduction and spread of saltcedar, or tamarisk, became common in most drainages in the Southwest after 1920. Saltcedar displaces native vegetation upon which certain species depend; it reduces the diversity of native shrubs and cottonwoods and transpires large quantities of water. Attempts to increase water yields by reduction of phreatophytes (such as saltcedar) have included root plows, dozer blades, various mowers and choppers, and chemical spraying. These treatments have declined significantly in recent years as a result of concerns about their efficacy and environmental impact.

Johnson (1978) estimates that only 10% of the original riparian habitat in United States remains, and that 6% is lost annually. Major losses resulted from drainage for conversion to agriculture; other causes include channelization for navigation and flood control, flooding caused by dam construction, and diversion of streamflow for irrigation. Alterations include grazing, timbering, road construction, mining, and other impacts. In Arizona, 95% of the woody riparian habitat has been lost or degraded since pre-settlement. In Utah, settlement patterns saw riparian areas converted to farmland, frequently hay fields. They continue to be threatened by water management activity, grazing, sand and gravel extraction, and development activities.

Wetlands

Specific western wetland problem areas identified by Tiner (1984) include the following:

- Estuarine wetlands of the U.S. Coastal Zone.
- Western riparian wetlands.

Wetlands in the Western Great Basin and Intermountain regions include riparian wetlands and shallow wetlands in pluvial lake basins. These shallow wetlands are often saline or alkaline as a result of high evaporation. Important large wetlands include the Bear River Marshes, UT, Malheur Lake Marshes, OR, Stillwater Marsh in the Carson Sink, NV, Tule-Klamath Basin in CA and OR, and the marsh systems of the California central valleys. Nesting habitat for Canada geese has been lost as much of the marshlands of the Great Salt Lake have been inundated with rising lake level (Thomas 1990). Important coastal estuary habitats include the large Gulf of California estuary and the fringing marshes along San Diego and Tomales Bays.

Aquatic Systems

The water area in *Western Grasslands and Deserts* is generally restricted to large bodies of water such as the Great Salt Lake (one-third of all water in the region), and the upper Missouri, Snake, and Colorado River systems.

Activities and Impacts Affecting Habitats

The *Western Deserts and Grasslands* has suffered extensive degradation and loss of rangelands through conversion to cropland; urban expansion; domestic and feral equine competition with indigenous populations for range resources; grazing-pressure effects from the introduction of shrub species to grasslands; and range management activities, including the use of herbicides and the exclusion of natural inhabitants (U.S. Forest Service 1989). Other activities negatively affecting rangelands include water management projects that dam or divert water supplies, mining impacts, and the use of remote rangelands as targets for waste disposal.

For example, in California more than 17 million ac of natural habitat have been lost through conversion to urban and agricultural land, including nearly 90% of riparian habitats in the Central Valley (California DFFP 1988). Major habitats that have lost significant acreages in the last 30 years include grasslands and coastal scrub. The use of grasslands for grazing also results in habitat loss and fragmentation, including excessive surface soil erosion on nearly 25% of western rangelands.

Grazing and water projects especially threaten riparian environments throughout the region. For example, overgrazing and phreatophyte control are destroying riparian vegetation in Arizona and New Mexico. Water diversions in the Central Valley and elsewhere have caused major losses of riparian and wetland habitats and are contributing to the declines of waterfowl along the Pacific Flyway.

Recreational use of off-road vehicles and military maneuvers are also degrading arid environments such as the Mojave. By one calculation, more than a half million ac have been disturbed by motor vehicles in California (California DFFP 1988). Fragile coastal dune habitats have also been damaged and eliminated by development, recreation, and introduced species.

The following activities result in the major impacts on habitats of concern in the *Western Grasslands and Deserts.*

IMPACTS ON HABITATS OF CONCERN IN THE *WESTERN GRASSLANDS AND DESERTS*					
	Land Conversion	Grazing	Mining	Water Management	Other
Riparian habitats	Residential development and construction of pipeline and transportation corridors	Degradation from domestic and feral ungulates	Moderate	Historical impact of impoundments and water diversions	Recreational use
Wetlands	Agricultural conversion	Moderate	Moderate	Historical impact of impoundments and water diversions	Minor
Deserts	Urban expansion	Degradation from domestic and feral ungulates	Moderate	Major impact of water diversions	Off-road vehicle use
Grasslands	Agricultural conversion	Degradation from domestic and feral ungulates	Minor	Minor	Minor
Woodlands and shrublands	Urban expansion	Moderate	Minor	Minor	Moderate

Land Conversion

To date, the most fertile soils within the *Western Grasslands and Deserts* have been converted to croplands; these same areas have historically supported the greatest abundance of wildlife (Mayer and Laudenslayer 1988). In addition, urban development has been a major source of rangeland conversions, reaching the highest urban densities at lower elevations with the majority of cities of 10,000 in population occupying areas formerly in grassland or scrub vegetation.

Urban and suburban expansion have converted large areas around the Los Angeles metropolitan area. In addition, some of California's fastest growing areas are in rural counties, including those with significant range resources. Rapid growth from the Sunbelt migration is now occurring around Las Vegas and other desert cities. In the Las Vegas area, the expansion of housing development has been facilitated by land trades with the Bureau of Land Management. Riparian areas in particular are under heavy pressure from development in New Mexico, Arizona, and Nevada. Pressure upon the land and local governments to convert open space to residential, commercial, and industrial uses to accommodate growth has been intense, and will continue to destroy rangeland habitats where population growth is most pronounced.

Conversion of rangelands to cropland will increase with the availability of ground water for irrigation (USDA Forest Service 1989). For example, sandy rangeland in Texas, Colorado, New Mexico, and Nebraska has already been converted to farmland (Sheridan 1981). Abandonment of these farms can lead to desertification if the ground water has been depleted. Areas of concern for desertification include the Challis Planning Unit in Idaho, the San Jaoquin Basin in California, the Gila, Santa Cruz, and San Pedro River Basins in Arizona, and the Sonoran and Chihuahuan Deserts in Southwest. In these arid and semiarid lands, land conversion to agriculture, grazing, and water management can cause groundwater overdraft, salinization of topsoil and water, reduction of surface water, high soil erosion, and destruction of native vegetation. Irrigation can also have adverse impacts on rangelands when poor drainage leads to waterlogged areas.

As with forest habitats, the spatial pattern and fragmentation of rangeland vegetation can negatively affect native fauna and ecosystem health. The loss of grassland habitat to agriculture is responsible for the decline in prairie birds, especially those requiring large continuous habitats, and is analogous to the reduction in old-growth forests and its obligate species. The upland sandpiper, bobolink, dickcissel, grasshopper sparrow, savannah sparrow, and Henslow's sparrow all declined by 90% between the 1950s and 1970s (Graber and Graber 1983).

Agricultural Impacts

The intensive use of agricultural land in certain areas of the *Western Grasslands and Deserts* poses additional stresses to habitats through cultivation and irrigation practices (NRC 1982). The use of fertilizers and pesticides, irrigation and drainage, double cropping, and increased field size all contribute to increased pollutant loads and severe impacts on habitats. Agricultural chemicals are toxic to many species and can negatively affect population levels, community composition, and ecosystem dynamics. Intensive cultivation practices (e.g., cotton agriculture in deserts) usually produce severe offsite impacts.

Grazing

Widespread devastation of rangeland resulted from uncontrolled overgrazing between 1880 and 1935, and the damage was undoubtedly amplified by the drought years of the 1930s (Branson 1985). The enactment of the Taylor Grazing Act of 1934 reduced grazing pressure at that time. With the advancement of range management science and the moist years following 1960, considerable improvement occurred in range vegetation. However, the USDA Forest Service (1989) reports 21% of its rangelands were still in "unsatisfactory" condition. The Bureau of Land Management (1989) reports that rangeland condition is 33% good or better, 38% fair, and 13% poor.

Overstocking and overgrazing have historically resulted in severe degradation and catastrophic flooding of rangelands. Undesirable and irreversible changes include replacement of grassland with creosote bush in the arid Southwest; replacement of native perennial bunchgrasses by Mediterranean annuals in California grasslands; and conversion of native vegetation in the Great Basin to an artificial balance of grasses and shrubs. Many national forest lands now contain different rangeland communities (e.g., invasion by Utah juniper into grass-shrub and replacement of grasses by big sagebrush).

The management of public land grazing is shared between the land management agency and the grazing permittee. Grazing permits are issued and allotments inspected for use, condition, and compliance by the management agency; actual management of the livestock and maintenance of improvements is the responsibility of the permittee. Attempts to reduce grazing allotments in national forests to allow improvements on lands in poor or fair condition has caused resentment among graziers. However, federal permit fees are only one-fifth the rate for private lands. As the acreage of private grasslands continues to decline with urban and agricultural conversion, there will be increased pressure on public lands.

Grazing poses the following threats to rangeland habitats (Cooperrider 1990):

- Competition with ungulates and small herbivores (e.g., desert tortoise) and limits on the populations of free-roaming pronghorn antelope, mule deer, elk, and bighorn sheep.
- Transmission of disease (e.g., dramatic diebacks in bighorn sheep with domestic sheep grazing).
- Loss of cover for birds.
- Spread of exotics and noxious weeds.
- Desertification, or serious degradation.
- The conversion of lands with sagebrush and pinyon-juniper to reseeded grassland for more forage.

Riparian Areas

The most severe impact in terms of supporting healthy ecosystems and native faunas on rangelands has been the loss of 70% to 90% of riparian areas to human activities (Ohmart and Anderson 1986). Losses of riparian areas have caused the endangerment of habitat-dependent species such as the Least Bell's vireo and likely will cause the extirpation of many species if the last remaining areas of individual types are lost (e.g., 10 species may become extinct if the cottonwood-willow association disappears). Johnson (1978) estimates that 6% of riparian areas continues to be lost annually. Historical loss estimates include 98% of riparian habitats in the Sacramento Valley of California, 95% in Arizona, and 90 to 95% in the Rocky Mountains Region. In Utah, settlement patterns saw riparian areas converted to farmland, frequently hay fields. They continue to be threatened by water management activity, grazing, sand and gravel extraction, and development activities.

Grazing is so ubiquitous in riparian ecosystems of the Southwest that only a few ungrazed sites exist (Szaro 1991). On average, the riparian zone is only 2% of a grazing allotment, but it produces 20% of the forage, and the cattle consume 80% of their forage from these riparian areas. Stream bottoms are natural concentration areas for livestock seeking succulent forage, shade, reliable water supply, and favorable microclimate. Only when access is limited by steep slopes are livestock absent from unfenced riparian areas. Grazing impacts riparian areas both by removing vegetation and by trampling. By affecting the spacing of plants, width of the riparian corridor, seedling establishment, and species composition, floristic diversity is often lower in grazed areas. Trampling increases soil compaction, erodes streambanks, decreases water quality, widens and shallows channels, and physically destroys vegetation (Kauffman and Krueger 1984). Riparian degradation causes accelerated runoff and erosion, downcut streambeds, lowered water tables, and desertification of the land. It has a negative impact on wildlife habitats and leads to declines in willows and native grasses. In addition, degraded riparian areas are more susceptible to upland inputs as healthy riparian areas can filter out upland degradation. While the condition of all rangelands has improved since 1980, riparian areas are in their worst historical condition.

Although the values and functions of riparian areas have been widely and severely impacted by cultivation, road building, mining, urbanization, logging, and damming of rivers, grazing has caused the most geographically extensive impacts (Chaney et al. 1990). Impacts of grazing on riparian areas include the following:

- Little vegetation to stabilize streambank and shade stream.
- Lowered water table and subsurface water storage.
- Reduced or absent summer flow.
- Warm water in summer and icing in winter.
- Poor habitat for fish and aquatics.
- Poor habitat for wildlife.
- Reduced amount and quality of forage.

Mining

Surface mining has severely degraded large areas of the *Western Grasslands and Deserts*. Surface deposits of minerals are extracted by removing successive layers of the terrestrial environment. Reclamation efforts have increased, but true restoration success is especially difficult in arid habitats. Establishment of vegetation is problematic even with fast growing nonnative species. Oil and gas development also pose severe risks to the pristine natural areas of the West. Exploration and production of both land and off-shore oil reserves are in direct conflict with many wildlife requirements. The substantial infrastructure required by mining activities also contributes to habitat degradation.

Water Management

The regulation and damming of streams are often performed to control flooding and drain land, resulting in the impoverishment of riparian vegetation (Szaro 1991). Dams and water diversion significantly change downstream flow regimes, levels of winter floodwater, dry-season flow rates, and riparian-zone soil moisture. Downstream areas lose pulse-stimulated responses while upstream areas are affected by water impoundment and salt accumulation. Native riparian plants are usually unable to colonize the shore of reservoirs because of the altered hydrologic regime. For example, high water levels are maintained much longer in reservoirs than in rivers and streams; changes in the level are more drastic; and the large winter/spring floods required for alluvial seedbeds (e.g., cottonwood) are eliminated.

Recreational Activities

The characteristics of riparian areas that attract wildlife and livestock also attract human recreation such as birdwatching, hiking, fishing, camping, hunting, trapping, picnicking, floating, boating, and river running (Carothers and Johnson 1975). These activities are increasing as leisure time, personal income, mobility, and pollution levels increase in the western United States. This will place even greater stress on these rare and abused ecosystems.

Military Activities

The large number of military training areas located in the *Western Grasslands and Deserts* results in major impacts on arid land environments. Both a reduction in vegetative ground cover and changes in species composition result from tracked vehicle activity and troop maneuvers (Diersing et al. 1992). There is a major shift from perennial warm-season grass (blue grama) to invading annual cool-season grasses following disturbance by tracked vehicles. This activity can also reduce densities of shrubs, trees, and succulent plants; the loss of juniper can exceed its ability to regrow.

Mitigations of Impacts

The conservation of habitats requires consideration of mitigations for the major activities impacting habitats of concern. In the *Western Deserts and Grasslands*, the primary habitat impacts are caused by the following:

- Grazing of riparian areas.
- Land conversion of riparian and wetland habitats.
- Urban conversion of desert and shrubland habitats.
- Mining impacts on arid lands.
- Water management impacts of diversion and damming on riparian and wetland areas.

It is likely that certain areas will see additional conversions to cropland or pasture, and that more open ranges will be fenced and thus restrict winter grazing by native ungulates. Increased irrigation will likely follow higher demand for water and adversely affect water tables and stream flow on rangelands. These and other activities will pose a complex of interrelated effects on habitats of concern and will require a holistic, ecosystem-level approach to mitigation. The effects of future management and mitigations on riparian areas will have the greatest impact on wildlife and native ecosystem health (NRC 1982).

Grazing

Future management of grazing on rangelands will determine whether range conditions worsen or improve from their currently degraded state (NRC 1982). In the past, range condition has been estimated by (1) forage production relative to a mythical average, and (2) production of livestock. Recently, some range managers have begun to base range condition on deviation from an ideal range or ecological climax. This and other improvements in range science provide for consideration of objectives beyond livestock production. For example, the widely used model of E.J. Dyksterhuis (1949) is based on reversible and gradual community change and is now viewed as inaccurate, as it does not incorporate threshold community shifts (Jahn 1991). The problem for habitat conservation is that the proportion of rangeland climax habitats has greatly decreased, similar to the case with old-growth forest. Although there remain disagreements over proper management methods, more effective use of ecological analyses of range condition will likely improve the management of rangelands.

Specific methods of mitigating grazing impacts on rangelands include the following (Branson 1985):

- Proper intensity and season of grazing.
- Practices that improve livestock distribution.
- Control of undesirable species using fire or other appropriate methods.
- Land-surface modification to retain soil moisture for forage production.
- Ecologically based management plans for each site using adequate field data.

Proper grazing management can restore the long-term productivity of most rangelands, but obstacles are grazing tradition, geographical extent of problem, and the difference between short-term costs and long-term benefits. Successful management requires that traditional intensive measures to increase forage be replaced by different management practices. For example, rest-rotation grazing can improve range conditions, while intensified chemical use and mechanical brush removal to improve forage will likely further degrade range habitats. In addition, fire can be used as a management tool to return pinyon-juniper areas to their previous savannah condition. As a rule, conversion to cattle from sheep requires more management as cattle use bottomland more intensely than sheep. Therefore, summer cattle use of desert ranges in an undesirable practice. Successful rangeland mitigation requires time, flexibility, commitment by graziers, and monitoring and evaluation.

Improvements in the condition of riparian areas will provide the greatest proportional benefit to rangeland integrity and functioning. The Bureau of Land Management (BLM) has plans for restoring 180,000 stream miles within 270 million ac of BLM lands to improve the functioning and status of 23.7 million ac of riparian/wetland systems to meet demands for protecting watersheds, restoring water quality, and enhancing conditions for fish, wildlife, livestock, and outdoor recreation (Jahn 1991).

Szaro (1991) argues strongly for an overall ecosystem approach to research and management of riparian areas. This includes the use of reference sites, a watershed (ecosystem) scale approach, and long time scale considerations (greater than 5 years). Mitigation of impacts to riparian areas should consider the following factors:

- Riparian floristic (plant species) diversity should take precedence over structural diversity (vegetation layers and patches) as descriptors of the habitat.

- Wildlife species depend both on floristic composition and on the relationship of riparian areas to animal movement patterns and migratory pathways.

- The distribution of riparian vegetative communities varies with topography and depends principally on elevation.

- Flooding and other natural disturbances are important to riparian systems. They contribute to their status as distinct and highly integrated pockets within other communities.

Successful riparian management requires unique solutions to the specific condition at each site (Chaney et al. 1990). However, general principles include the following:

- Include riparian areas in separate pastures with separate objectives and strategies.
- Fence or herd stock out of riparian areas to let vegetation recover.
- Control the timing of grazing (1) to keep the stock off streambanks they are most vulnerable to erosion, and (2) to coincide with the physiological needs of plants.
- Provide more rest to the grazing cycle to increase plant vigor or encourage more desirable species.
- Limit grazing intensity.
- Change from cattle to sheep to get better animal distribution through herding.
- Permanently exclude livestock from high-risk and poor-recovery areas.

Land Conversion

Effective mitigation of land conversion activities can sometimes be obtained only by avoiding impacts on rare or unusual habitat types. Rarely, if ever, is restoration or compensation an adequate mitigation for the loss of these habitats. In these cases, mitigation is a siting issue, where construction and degrading activities are located a distance from the habitats of concern. The habitat is adequately preserved if all possible impact scenarios are accounted for. Barring this solution, effective management measures must be implemented to ensure the protection of the habitats of concern.

In the case of unique riparian or wetland habitats, hydrological and contamination concerns are especially important. Construction or resource management activities require the use of sediment filter strips and other means of intercepting offsite contaminants. Road building and structural "improvements" must not result in altered hydrological regimes. Desert habitats are especially vulnerable to mechanical disruption by vehicles and machinery. Where rare plant types exist or where habitats are unstable (e.g., sand dunes), recreational access may have to be limited. These mitigations can be best implemented by creation of a regional land-use plan (through a coordinating council like the Waterfowl Flyway Council) and landowner incentives like the Conservation Reserve Program.

Conversion to agricultural land is a special concern in rangelands with increasing irrigation potential. Land conversion to agriculture can cause groundwater overdraft, salinization of topsoil and water, reduction of surface water, high soil erosion, and destruction of native vegetation. Mitigations include more conservative irrigation techniques and improved drainage systems. Soil conservation techniques vary from windbreaks to contour plowing, stripcropping, rotation of crops, conversion to grass, and/or minimum tillage.

Amelioration of impacts from land conversion to transportation uses requires special mitigation measures. As with all land conversion, the construction of highways and power-line corridors is primarily a siting issue. Avoidance of sensitive habitats may be accomplished by modifications to the route design, and the extent of disturbance can be limited by careful construction practices. However, fragmentation of the larger area is unavoidable in the case of land conversion to transportation corridors. Many structural mitigation measures can be used to lessen the impact on animal movement across transportation routes. Primarily, these include the construction of fences and underpasses. The goal of these structural measures should be to mimic the natural movement and migration patterns of the affected species.

<u>Mining</u>

Mitigation of mining impacts involves siting issues, technological solutions to eliminate contamination, and restoration programs. The major mitigations for oil and gas extraction and production are the proper sitings of rigs, reserve pits, processing facilities, and roads where they will have minimal impacts on habitats of concern. Most important for coal and mineral mining is the siting of mining operations and tailing ponds to avoid habitats of concern, wetlands, riparian areas, and recharge areas. Specific mitigation measures depend on the type of mining and the specific process causing impacts. It is generally best to minimize the area affected as it is unlikely that even the disrupted soils and sediments can be restored. In addition to minimizing the area disturbed, activities should be timed to avoid disturbing nearby plants and animals during crucial periods of their life cycle.

Possible mitigation measures for mining operations include the following (SAIC 1991a, 1991b):

- Design of mine entrances and workings to minimize future mine drainage.
- Runon and runoff control measures such as berms and ditches.
- Adequate depth and lining of pits for containment of muds and leachate.
- Elimination of migration of fluids through casings and dewatering.
- Separation of wastes and contaminated soils with proper disposal.
- Treatment of leach heaps and neutral or acidic wastewaters to reduce the load of cyanide, nitrates, and heavy metals.
- Closure planning that addresses hydrology, geochemical controls, treatment, and restoration.
- Nets or other covers over process ponds.
- Maintenance of an anaerobic environment in the tailing pile during periods of inactivity.
- Secondary containment of tanks and contingency plans for sudden or catastrophic releases.
- Backfilling and sealing of the mine workings during mine reclamation/closure.
- Recycling of process water, smelter slag, and air pollution control dust.
- Monitoring and elimination of discharges to surface water, groundwater, soils, and air.

- Replenishment of surface and ground waters with treated effluents.

- Road closure and reclamation (following recontouring) with revegetation of native species.

Although the reclamation of mined lands is often unsatisfactory for ecological habitat restoration, reforestation with native trees has been demonstrated (Plass 1975) and would serve to reduce the abundance of nest parasitic brown-headed cowbirds and restrict their access to mature forest.

Wetlands

Mitigation of wetlands destruction and degradation is the subject of a growing body of literature (Kusler and Kentula 1989). Restoration and mitigation banking concepts are still being evaluated as effective mitigation measures for direct wetlands alterations.

Military Activities

Mitigation of the impacts of military activities on habitats has only recently received attention. The Army Corps of Engineers' Construction Engineering Research Laboratory in Champaign, IL, is developing a Land Condition-Trend Analysis (LCTA) Program (Diersing et al. 1992) as a comprehensive means of matching military training mission objectives with effective natural resource management. If such a plan is instituted, it is likely that careful coordination of the siting and timing of training operations will dramatically reduce habitat impacts. An awareness of the ecological consequences of specific activities is essential to effective mitigation. The following general mitigation measures apply the primary impacts of military activity.

- Timing and siting of operations - The noise and disturbance associated with aircraft flights and large troop maneuvers cannot be eliminated. However, sensitive environments can be avoided, and operations can be timed to avoid critical nesting and migratory periods.

- Calculation of allowable use for tracked vehicles - Tracked vehicle movements are a major cause of habitat degradation. Vegetation destruction and soil erosion and compaction are the primary impacts. Precise equations can be developed that estimate sustained tracked vehicle use based on physical properties of the environment, vegetative cover, and changes in vegetative cover caused by the passage of tracked vehicles. For example, tracked vehicle use should be restricted to all-weather roads when possible.

- Fire suppression during artillery practice - Fires created by artillery pose a major problem in arid environments. Rapid identification and suppression by helicopter can virtually eliminate the spread of large-scale fires.

Guidelines for Reviewers

Reviewers of environmental impact assessments will find this document useful if they follow the steps laid out in the introduction:

1. Review the status and trends of habitats in the region.

2. Identify the habitats of concern.

3. Link the activities involved with impacts to these habitats of concern.

4. Devise appropriate mitigations for the impacts.

Each reviewer can then determine the adequacy of the environmental impact assessment in question and recommend modifications to enhance its effectiveness.

In identifying the habitats of concern, the reviewer should supplement the information in this document with detailed locational information on the abundance and distribution of habitats within the region of interest, and with any historical information on the extent and quality of these habitats. Most important, ~~the reviewer~~ should characterize ~~the~~ habitats in terms of their ecological values (e.g., use of wooded wetlands by migratory ~~waterfowl~~).

In considering the links between activities and habitats, the reviewer should look beyond direct impacts to indirect and subtle effects, including cumulative impacts, interactive and synergistic impacts, and scale-dependent impacts (e.g., effects of fragmentation on ecosystem integrity and species home ranges).

In devising possible mitigations, the reviewer should follow the seven principles for habitat mitigation repeated below. The reviewer should also determine whether adequate assurances have been given that the mitigations proposed will be completed.

1. Base mitigation goals and objectives on a landscape-scale analysis that considers the needs of the region.

2. Mimic natural processes and promote native species.

3. Protect rare and ecologically important species and communities.

4. Minimize fragmentation of habitat and promote connectivity of natural areas.

5. Maintain structural diversity of habitats and, where appropriate, species diversity to promote the natural variety of the area.

6. Tailor management to site-specific environmental conditions and to the unique impacts of the specific degrading activity.

7. Monitor for habitat impacts and revise mitigation plans as necessary.

Finally, the reviewer should consider the proposed activities and mitigations in the context of relevant regional program goals and objectives (e.g., whether the outcome of the project will be in accordance with principles set out by regional planning commissions such as those established for southern California).

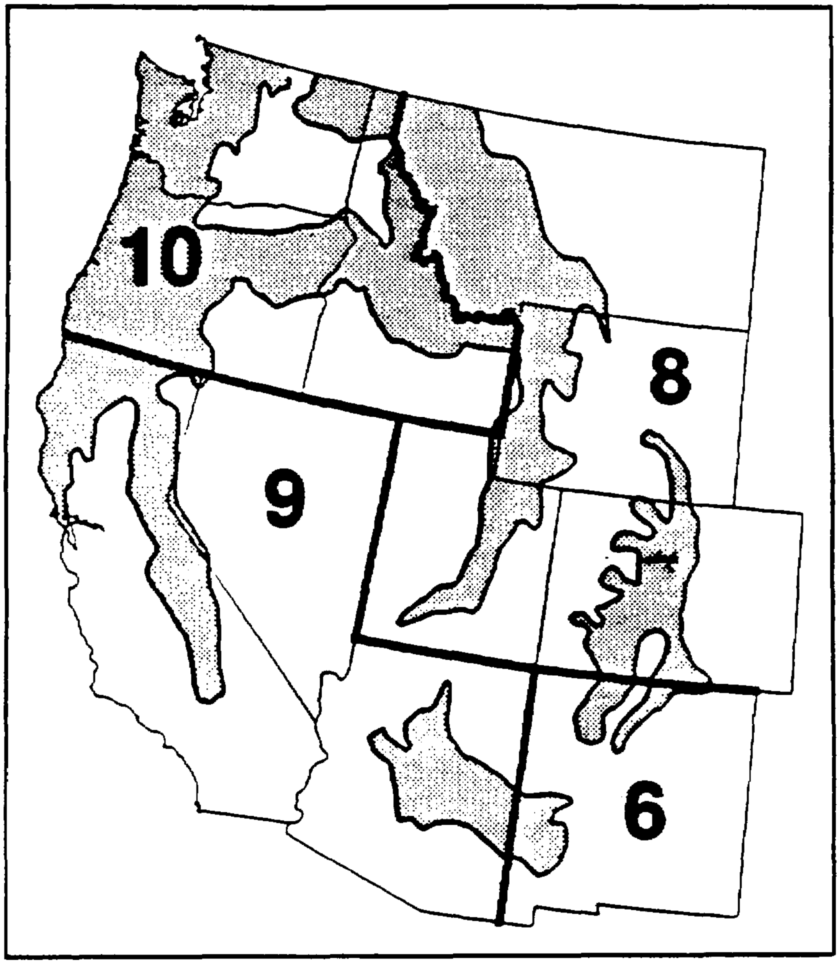
Contacts and Information Sources

When considering habitat conservation issues in an environmental impact assessment for the *Western Deserts and Grasslands*, the reviewer should consult the following organizations and individuals for information on habitat impacts and mitigations:

State Natural Heritage Programs
U.S. Fish and Wildlife Service, Regional and Area Offices
State Fish and Game Departments
University and Research Programs
Herbaria and Museums

Patrick Bourgeron, Regional Ecologist, The Nature Conservancy
R. Langley, Associate Director, Fish and Wildlife Enhancement, U.S. Fish and Wildlife Service

Western Forests Habitat Region: *Western Forests*



Geographical Description of Region

The Western Forests Habitat Region, *Western Forests*, contains parts of 11 states. The region includes parts of Washington, Oregon, California, Arizona, New Mexico, Colorado, Wyoming, Utah, Nevada, Montana, and Idaho. Parts of EPA Regions 6, 8, 9, and 10 are included. The accompanying map indicates the boundaries of this habitat region and the states it comprises.

The *Western Forests* comprises 12 ecoregions (Omernik, 1987). The vegetation of the *Western Forests* includes a wide range of forest types, including spruce, cedar, hemlock, cedar hemlock, Douglas fir, redwood, silver-fir, western spruce, mixed conifer forest (fir, pine, Douglas fir), red fir, lodgepole, subalpine forest, western ponderosa pine, grand-fir, alpine meadows (bent grass, sedge, fescue, needlegrass), Arizona pine, pinyon woodland, Southwestern spruce, and a mosaic of Oregon oakwoods. The land use pattern is predominantly forest and woodlands that are grazed and ungrazed, pasture croplands, and croplands with some interspersion of pasture, woodlands, and forests.

Habitats of Concern

The *Western Forests* contains many habitats of concern; the most obvious fall into four general categories: old-growth conifer forests, remnant hardwood forests, alpine communities, and riparian and aquatic systems. The principal habitats of concern most at risk in the *Western Forests* are listed below.

PRINCIPAL HABITATS OF CONCERN IN THE *WESTERN FORESTS*

1. Old growth conifer forests
   - Douglas fir of Pacific Northwest
   - ponderosa pine east of the Cascades
   - ponderosa pine in Arizona and New Mexico
   - redwood forest in California
   - old growth in northern Rocky Mountains

2. Remnant hardwood forests
   - California oak woodlands
   - Oregon oakwoods

2. Alpine communities
   - montane grasslands
   - alpine talus and barrens

3. Riparian, wetland, and aquatic systems
   - riparian forest
   - salmon fishery habitat

Habitat Values and Trends

The western United States contains a large area of forested land, including the last substantial areas of virgin forest (excluding Alaska). Timber harvesting came to the West with the settlement era after most of the East had already been logged. The three major regions of western forests are the Rocky Mountains, California, and the Pacific Northwest.

Rocky Mountain Forests

In the Middle and Southern Rocky Mountains, intensive exploitation of forest timber began when railroads opened up the region, producing lumber mills in 1870 (Barrett 1980). Fire also played an important role in this region, promoting lodgepole pine at the expense of Douglas fir. Logging came later to the Northern Rocky Mountains, where the forests of Idaho and western Montana represent the largest area of contiguous forest in the United States with more than 80% of the land forested.

Agricultural settlement increased rapidly after the Civil War, reaching into the fertile grasslands and open timbered foothills. Farmland extension is currently slow but continuing into the forest area. About half of the forest area is grazed.

Current forestry efforts are directed at the conversion of old-growth and high-graded stands to commercial timber harvesting. Although white pines forests were intensively logged between 1910 and 1925, old-growth forests still predominate over much of the Northern Rocky Mountain Region. A total of 138 million ac of forest occur in the Rocky Mountain Region, most in pinyon-juniper woodland (47 million ac of dry plateaus and broken tablelands), Douglas-fir (18 million ac), fir-spruce (16 million ac), ponderosa pine (16 million ac), and lodgepole pine (15 million ac) (USDA Forest Service 1989). In recent decades, a modest, steady decline in forest area has occurred as a result of clearing for roads, urban development, powerline rights-of-way, and surface mining. Substantial areas in Montana, Idaho, and Colorado have been converted to homesites. Data indicate that forest ecosystem types that have declined since 1963 include western white pine (89%), larch (35%), lodgepole pine (29%), ponderosa pine (27%), and western hardwood (19%). In the future, forest area is expected to remain stable as timber harvesting lands decrease and conversions to urban uses increase.

The Rocky Mountain region is a highly dissected series of peaks and ridges containing both forests and rangeland (see Western Rangelands Habitat Region). Even within forested areas, many unusual habitats exist, including old-growth spruce/skunk cabbage, acid shale ponderosa pine communities, intermountain bunchgrass, and various alpine and subalpine communities. Many of these are uncommon and isolated, representing especially vulnerable habitats in this region.

California Forests

California is second only to Alaska in total forest area; forest area constitutes 40% of the state, or 40 million ac (Barrett 1980). Since 1953, the total commercial forest area in California has decreased by about 1 million ac because of grazing development, roads, construction of reservoirs and power lines, urban expansion, and park and wilderness dedication. The six major habitat types include redwood, mixed conifer, true fir, ponderosa pine, California oak woodland, and California chaparral. Although the state has a long history of industrial use of forest, efforts are under way to restrict timber harvesting throughout the state.

Losses of forests and woodlands have been less than 1% per year over the last decade and are caused principally by urbanization and construction of roads and reservoirs (USDA Forest Service 1989). However, the condition of forests has been greatly affected by logging, which has reduced the number of trees by 55% and changed open stands of large trees to dense stands of small trees. Forest composition has changed; hardwoods have replaced coastal conifers, while white fir and incense-cedar have replaced pine in the interior. Originally 74% of forest was mature or old growth and 13% was in sapling or saw timber stages. Now nearly 40% of mature stands have been cut and are in the sapling stage. Predictions are that about 11% of timberland will be reserved for mature stands (Raphael et al. 1988). Air pollution, both acid deposition and smog, also have caused extensive damage to these forest ecosystems, especially to the susceptible granitic watersheds and Southern California forests (California Department of Forestry and Fire Protection 1988).

Pacific Northwest Forests

The Pacific States, excluding California, comprise about 50 million ac of forest. Major types include western hemlock-sitka spruce, coastal Douglas fir, true fir-mountain hemlock, mixed conifers of southwestern Oregon, mixed pine-fir of eastern Oregon and Washington, and northwestern ponderosa pine (USDA Forest Service 1989). Since 1963, many forest ecosystem types have declined: western white pine (99%), redwood (31%), ponderosa pine (26%), Douglas fir (20%), and lodgepole pine (17%).

The Pacific Northwest rainforest (principally spruce, hemlock, and fir) constitutes one of the most productive forest regions in the world. The western areas of Washington and Oregon are 80% forested, and the eastern portions of these states are 35% to 40% forested. Large-scale settlement began in the Pacific Northwest during the middle of the 19th century. Agriculture was restricted to river valleys and the steppe vegetation of the East, but adjacent forested areas were used extensively for grazing of both sheep and cattle. Timber harvesting increased with the advent of the California Gold Rush and has continued to be a major industry ever since. Forest use west of the Cascades started along waterways and progressed inland onto steeper slopes as logging technologies improved. Virgin timber is still being cut on the higher slopes of the Olympics and western slopes of the Cascades, but the age classes of the second forest follow the original, regional pattern of harvesting. Clearcut logging has been almost universal west of the Cascades with partial cut logging used to the east (Barrett 1980).

The Olympic Peninsula of Washington contains one of the best examples of old-growth forests remaining in the United States. Of the 390,000 ac of old growth existing in 1940, only 94,000 remained in 1988 (Morrison 1990). Although sitka spruce and western hemlock covered more than 1 million ac before European settlement, logging and human-caused fires have reduced the area by 97%. Additiona' ecological zones include Douglas fir, pacific silver fir, mountain hemlock, subalpine fir, and alpine. In both Oregon and Washington, the most obvious change in forest cover over the last 10 years has been the reduction in area of old-growth forests by logging. Major impacts in both states have been clear cutting, road building, edge effects, fragmentation, and human fires, as well as disease and pest mortality in eastern Washington.

Morrison (1988) assessed the amount and condition of ecological old-growth conifer forest that still exists on 6 of the 12 westside national forests in the coastal region of Oregon, Washington, and northern California and estimated the amount of old growth that will remain in 5 years if present policy continues. The results predict that old growth covers less area and is being lost more rapidly than is claimed by the U.S. Forest Service. Factors contributing to the vulnerability of old-growth forest in the Northwest include the following:

- Nearly all of the old growth on private lands in the Pacific Northwest has been logged.
- Only 31% of the remaining old growth is in designated wilderness areas.

Based on 1984 maps (Flather and Hoekstra 1989), the average number of endangered and threatened species per county is 5.6 for the *Western Forests*, among the highest in the nation. The following listing of Pacific Northwest forest types illustrates some of their characteristic ecological values:

Douglas-fir - dense overstory forest of ancient trees supports important plants such as epiphytes and yew, and rare species such as spotted owl and marbled murrelet; forest openings and early seral stages support elk, grizzly bear, moose, blue and ruffed grouse, mammalian predators such as mountain lions and bobcats, and endangered American peregrine falcon.

Fir-spruce and hemlock-Sitka spruce - dense canopy forest with little understory but interspersed with meadows or stream bottoms with willows and aspens; support moose, elk, wolverine, lynx, black bear, mountain lion, and some grizzly bear.

Ponderosa pine - historically, fire kept habitat open and park-like with ground cover of grasses, sedges, and forbs; supports black bear, mule deer, elk, and mountain lion.

Lodgepole pine - supports moose, elk, wolverine, lynx, black bear, mountain lion, coyote, and some grizzly bear.

Redwood - dense overstory forest of small geographic extent in California and Oregon; supports elk, mountain lion, bobcat, and black bear.

Western hardwoods - 50% or more of coast live oak, canyon live oak, blue oak, valley oak, interior live oak, or aspen; in California supports mule deer, California quail, mountain quail, skunk, and endangered San Joaquin kit fox.

Pinyon-juniper - often adjacent to sagebrush on dissection of western basins and mountains; supports mule deer, mountain lion coyote, bobcat, jackrabbit, numerous birds.

Alpine - above timberline in Rocky Mountain and Pacific Coast regions; consists of grasses, grasslike species and forbs; includes lakes and ponds with endemic trout; supports pika, pocket gopher, yellow-bellied marmot, mule deer, elk, mountain sheep, and ptarmigan.

Riparian and Wetland Areas

The original amount of wetland area in the Rocky Mountain Region has been decreased by one-third since widespread settlement began (Windell et al. 1986). The Rocky Mountains comprise a relatively small area of wetlands, but a wide variety of wetland types, ranging from intermountain basins to alpine tundra. Much of the impact results from the concentration of human population within certain Rocky Mountain areas. Population tends to be sparse in the high plains, heavy along the junction between the plains and mountains, and moderate in the mountains along narrow valley floodplain corridors. The heaviest development is concentrated along water courses.

Development along water courses has dramatically reduced the area of wetlands in the Pacific States. As in the Rocky Mountain Region, many Pacific States wetlands occur in rangeland environments rather than forests. However, many wetlands do occur in the *Western Forests*, including the large estuaries of San Francisco and Puget Sound and the forest wetlands along the north coast of Washington. Perhaps of even greater importance in the *Western Forests* are riparian areas. These forest zones provide essential habitat for many forest species, connect forest to wetland areas, and provide filtering and transport of nutrients for aquatic systems. The traditional use of riparian areas for access to timber harvesting and transport of logs has severely degraded riparian areas in the *Western Forests*.

Aquatic Systems

Approximately 6 million ac of water area occur in the vast Rocky Mountains. About 4 million ac of water area occur in the Pacific States, including coastal waterways such as Puget Sound and Strait of Juan de Fuca, Crater Lake, and rivers such as the Columbia and Willamette. Incomparable salmonid fisheries were once characteristic of the *Western Forests*. Timber harvesting practices and development on major rivers, especially damming for hydropower and irrigation diversion, have dramatically reduced fishery habitat and salmonid abundance.

Activities and Impacts Affecting Habitats

The major sources of degradation and loss to terrestrial environments in the *Western Forests* are timber harvesting practices and mining. Land conversion and water management activities also affect both terrestrial and aquatic systems. The ecologically rich old-growth forests of the Pacific Northwest are under intense logging pressure as private old-growth lands are eliminated. The total area of old growth has declined by 80%, and the remaining forests are being fragmented and degraded. This issue represents one of the country's most intense conflicts of natural area preservation and resource exploitation.

In addition to timber harvesting, mining and oil and gas development pose risks to the pristine natural areas of the Northwest. Gold mining is causing habitat degradation in Washington. Pressure upon local governments to convert open space to residential, commercial, and industrial uses to accommodate growth have been intense, and have also been responsible for the loss of wildlife habitat in the area. Losses have been most severe where the effects of urbanization and population growth are most pronounced. California habitats that have lost significant acreages in the last 30 years include foothill oak woodland, closed-cone pine-cypress, and redwood forests. Much of the development in the next decade will occur on hardwood forest lands of California.

The following activities result in the major impacts on habitats of concern in the *Western Forests*.

IMPACTS ON HABITATS OF CONCERN IN THE *WESTERN FORESTS*				
	Land Conversion	Timber harvesting	Mining	Water Management
Old-growth forests	Minor	Clearcutting and forest fragmentation	Moderate	Minor
Remnant hardwood forests	Urban development	Moderate	Minor	Minor
Alpine communities	Resort and recreational development	Minor	Moderate	Minor
Riparian, wetland, and aquatic systems	Residential development in river bottoms and construction of pipeline and transportation corridors	Major impacts of erosion and sedimentation	Major	Major impacts of damming and water diversion

Timber Harvesting

Old-growth forests are of special concern for habitat conservation. Not only do these sensitive terrestrial environments contain unique assemblages of species but they are also under intense timber harvesting pressure. The only significant remaining area of old-growth forest is the conifer forest of the Pacific Northwest. Less than 5 million ac of the original 15 million ac of old growth in western Washington and western Oregon remain. Some view the altered landscape of the Olympic Peninsula in western Washington due to timber cutting as the most drastic ecological disturbance of the last 10,000 years (Morrison, 1990). Less than 20% of the original old growth on the peninsula remains, and entire ecological associations of plants and animals that once dominated lower elevations on the peninsula are now rare. Ancient forests of the Pacific Northwest have been so fragmented by roads and logging that the viability of the old-growth ecosystem is in question.

Forests serve many important ecosystem functions that can be lost or degraded by timber harvesting practices (Norse 1990a). For example, forests are naturally efficient regulators of water-flow levels through the retention of surface run-off during high precipitation periods and the maintenance of moisture levels during low precipitation periods. Forest stabilization of soils prevents increases in sediment loads and maintains water purity for aquatic habitat and human uses. In the Klamath Mountains of southwestern Oregon, erosion rates in roaded areas averaged more than 100 times higher than on undisturbed sites, and erosion caused by logging alone averaged 6.8 times higher than on undisturbed sites (Dyrness 1975). In northern California over a 9-year period, stream sediment in a developed watershed was more than 80% higher with road building and 275% higher with logging and roads than in a similar, undisturbed watershed. Forests also serve to retain nutrients within the ecosystem by a complex process of litter accumulation ahd decomposition. Logging often destroys the nutrient retention ability of the soils and has been implicated in failures to achieve forest regeneration.

Fragmentation of habitat is another severe impact of timber harvesting on forests. As roads and clearcuts are placed in virgin forest, landscape fragmentation increases and the natural buffering of extremes in temperature, drought, snow pack, and wind decreases. As a result, blowdowns, fires, insect and disease infestations, snag cutting, and salvage logging increase. Approximately 60 ac of old growth are destroyed or altered for each new 25-ac clearcut in unfragmented old growth as a result of deleterious edge effects; for every mile of road built in unfragmented old growth, approximately 97 ac of old-growth forest are altered by edge effects (Morrison 1988).

Land Conversion

Land conversion in the *Western Forests* has the greatest impact on the remnant woodlands at the edge of urban centers and on the forest valleys along river courses. The U.S. Forest Service (1989) projections over the next 50 yr indicate a loss in forest area of 8 million ac with the conversion to urban and developed uses in the Seattle-Tacoma areas, numerous areas in California, and the mixed forest-urban zones of Oregon. Conversion of both uplands and wetlands has a profound effect on the natural communities in the West. In recent years, the expansion of populations into formerly pristine areas is fragmenting forest through industrial and residential development. Rural areas are also suffering from "spin-off development" associated with highway development.

Impacts on Riparian and Wetland Areas

In addition to the conversion of lands along water courses, riparian and wetland areas of the *Western Forests* face threats from other offsite and onsite activities. The primary impacts to wetlands include the following:

- Recreation and other development (especially vacation houses and resort facilities).
- Drainage and filling for buildup and parking areas (impact of cumulative effects).
- Dewatering, diversion, and irrigation (there are many transbasin diversion systems in the Rocky Mountains).
- Forest clear-cutting and channelization (causing erosion, faster snowmelt, reduced water retention, and nutrient loading downstream).
- Mineral mining (aquifer draw down, channelization, stream diversion, acid and alkaline mine drainage, waste disposal sites and tailing areas, erosion and sedimentation).
- Sand and gravel mining (expected to triple or quadruple by the year 2000).

- Road and railroad access (construction of roads, villages, and towns along medium to large streams).

- Dams and reservoirs (decreasing the acreage of riverine, riparian, and wetland systems).

Impacts on Aquatic Systems

Aquatic resources, especially the anadromous fisheries of the Pacific Northwest, are also suffering severe declines. The complex of dams on the Columbia River kill approximately 93% of young salmon and have contributed to the listing of the sockeye and chinook salmon as threatened. Recovery plans for these and other fish species will have large-scale ramifications on water management and human industry planning for the region (Weisskopf 1991).

Mitigations of Impacts

The conservation of habitats requires consideration of mitigations for the major activities impacting habitats of concern. In the *Western Forests*, the primary habitat impacts are caused by the following:

- Timber harvesting and fragmentation of old-growth forests.
- Land conversion of remnant hardwood forests and alpine communities.
- Mining impacts on forests and aquatic systems.
- Water management impacts of diversion and damming on rivers.

Management of the combined effect of these activities on sensitive habitats requires a holistic, ecosystem-level approach. The new interagency efforts to manage the Greater Yellowstone Ecosystem in Montana and Wyoming (approximately 20 million ac, 69% publicly owned by five federal agencies) is the premier example of an integrated approach to ecosystem management (Jahn 1991). In particular, the approach pays special attention to the needs of wide-ranging species such as elk and grizzly bears. It emphasizes the need to look at the landscape scale (not institutional boundaries) for the implications of habitat value and modification.

Timber Harvesting

At a minimum, the production of commercial wood products from an area must not exceed the sustainable level if the ecological integrity of a forested area is to be maintained. Where sensitive forest types exist, logging may be completely prohibited or constrained to specific methods to prevent habitat loss or degradation. In other areas, more extreme harvesting methods may be allowed or prescribed to establish or maintain desired forest conditions. Acceptable methods will vary according to local forest ecology and the desired future condition of the site. Analysis of harvesting techniques must be based upon an analysis of the structure and diversity of the forest canopy, midstory, and understory.

A recent directive of the Chief of the U.S. Forest Service acknowledges this fact and points out that clear cutting is acceptable only when needed to replicate natural ecological processes. Selective cutting can preserve forest structural diversity, the primary determinant of wildlife habitat (Harris et al. 1979). However, it can reduce horizontal diversity (NRC 1982). The harvesting technique employed must be based upon sound silvicultural prescriptions and demonstrate its capability to maintain vertical diversity (foliage height diversity), horizontal diversity (interspersion, edge, juxtaposition, patchiness), and a mixture of live and dead wood. Specific timber harvesting operations should be designed to preserve the structure and diversity of the natural forest habitat.

An important component of selective cutting should be the preservation of standing dead trees. Many forest birds nest, roost, or forage for invertebrates in standing trees with decayed wood. These cull trees are usually the first focus of forest-thinning operations to the detriment of the birds. Breeding bird abundance declines rapidly following a clear cut, and the species composition continues to change for 10 to 15 years (DeGraaf 1991). However, if trees with cavities are saved, many of these species can successfully forage on sound boles. About one large cavity or den tree per 2 ha is required for population of large species such as wood ducks; this requires harvest rotations of 100 to 125 years (although rotations of 65 years produce trees large enough for species nesting in smaller cavities).

Timber harvesting practices modified to reduce the impacts of simplification must also address fragmentation. The setting aside of undisturbed tracts will not achieve viable populations of the larger, wider-ranging species. Some species require specific habitat conditions; others require particular arrangements of several communities. Therefore, a successful faunal conservation strategy must emphasize the landscape configuration, not just the structural content of the communities themselves.

Responding to the "biodiversity crisis," the U.S. Forest Service is moving toward an ecosystem approach to forest management (Bob Szaro, personal communication). Recent forest management plans have incorporated tenets of the "New Forestry" espoused by Jerry Franklin. These progressive plans require the rigorous implementation of ecological management practices to maintain forest productivity and preserve the functioning of sensitive forest components such as old-growth or late-successional forests. Effective mitigations for habitat conservation in forest management require specific management measures at the site, watershed, and landscape levels. For example, the location and size of timber harvests should be planned to minimize reduction of core area of mature forest (e.g., harvest only alternate basins until regrowth). Maintenance of mature-forest stands in managed landscape can be achieved by extending rotation (beyond 80) to 150 to 200 years, by leaving some stands unharvested for old growth, and by linking stands. Landscape-scale considerations include the provision of buffer zones and habitat corridors as discussed in the introduction to this document. Management measures recommended for conserving habitat within managed forests include the following:

- Minimize the construction of new roads and close roads not in use either permanently or seasonally.

- Use best management practices (BMPs) such as filter strips to minimize erosion during harvesting or road construction.

- Maintain 100-ft riparian zones with adjacent feathered transition zones to buffer edge effects.

- Restrict harvesting operations to periods when the ground is either dry or frozen.

- Maintain site productivity by retaining large woody material and minimizing mineral soil exposure and compaction during harvesting.

- Manage for natural disturbance patterns to maintain natural openings and successional-stage composition.

- Maintain connections between blocks of interior forest, especially old growth.

- Provide for the protection of special areas, including cliffs, caves, taluses, riparian areas, and old-growth stands.

- Maintain the structural integrity and the native variety of the forest by managing for the natural composition of the following components: vegetative types, seral stages, tree types and sizes, standing dead trees and down material, tree snags, and cavity trees.

The preservation of old-growth forest in the Pacific Northwest has been the focus of intensive scientific study. For example, the report of The Scientific Panel on Late-Successional Forest Ecosystems provides a model of alternatives of forest management for preservation of ecosystems and wildlife (Johnson et al. 1991). Using the spotted owls as an indicator species, the panel derived the following recommendations for mitigating the impact of timber harvesting on late-successional/old-growth forest in the Northwest:

- Late-Successional/Old-Growth (LS/OG) areas should be protected as habitat conservation areas (HCAs). Blocks suitable to maintain 20 pairs of owl should be not more than 12 miles apart. Areas between these blocks must follow the 50-11-40 rule: 50% of forest must have an average tree diameter of 11 inches and canopy closure of 40%. Areas with additional owls may be added to the HCAs to meet the goal of preservation.

- Provisions for watersheds and fish include major reductions in road mileage and road drainage improvements, as well as extended logging rotations. "Problem" roads would be improved or removed, and unstable soils would remain unroaded.

- Riparian management will include no-harvest areas of varying width (1/4 mi to 50 ft depending on the value of the stream).

In a series of alternatives (from high timber harvest to LS/OG and watershed/fish emphasis), the Panel found that "current forest plans do not provide a high level of assurance for maintaining habitat for old-growth-dependent species." No alternative provides abundant timber harvest and high levels of habitat protection for species associated with late-successional forests.

Land Conversion

Effective mitigation of land conversion activities can sometimes be obtained only by avoiding impacts on rare or unusual habitat types. Rarely, if ever, is restoration or compensation an adequate mitigation for the loss of these habitats. In these cases, mitigation is a siting issue, where construction and degrading activities are located at a distance from the habitats of concern. The habitat is adequately preserved if all possible impact scenarios are accounted for. Barring this solution, effective management measures must be implemented to ensure the protection of the habitats of concern.

In the case of unique woodland or wetland habitats, hydrological and contamination concerns are especially important. Construction or resource management activities require the use of sediment filter strips and other means of intercepting offsite contaminants. Road building and structural "improvements" must not result in altered hydrological regimes. Where rare plant types exist or where habitats are unstable (e.g., riparian areas), recreational access may have to be limited. These mitigations can be best implemented by creation of a regional land-use plan (through a coordinating council like the Waterfowl Flyway Council) and landowner incentives like the Conservation Reserve Program.

Mining

Mitigation of mining impacts involves siting issues, technological solutions to eliminate contamination, and restoration programs. The major mitigations for oil and gas extraction and production are the proper sitings of rigs, reserve pits, processing facilities, and roads where they will have minimal impacts on habitats of concern. Most important for coal and mineral mining is the siting of mining operations and tailing ponds to avoid habitats of concern, wetlands, riparian areas, and recharge areas. Specific mitigation measures depend on the type of mining and the specific process causing impacts. It is generally best to minimize the area affected as it is unlikely that even the disrupted soils and sediments can be restored. In addition to minimizing the area disturbed, activities should be timed to avoid disturbing nearby plants and animals during crucial periods of their life cycle.

Possible mitigation measures for mining operations include the following (SAIC 1991a, 1991b):

- Design of mine entrances and workings to minimize future mine drainage.
- Runon and runoff control measures such as berms and ditches.
- Adequate depth and lining of pits for containment of muds and leachate.
- Elimination of migration of fluids through casings and dewatering.
- Separation of wastes and contaminated soils with proper disposal.
- Treatment of leach heaps and neutral or acidic wastewaters to reduce the load of cyanide, nitrates, and heavy metals.
- Closure planning that addresses hydrology, geochemical controls, treatment, and restoration.
- Nets or other covers over process ponds.
- Maintenance of an anaerobic environment in the tailing pile during periods of inactivity.
- Secondary containment of tanks and contingency plans for sudden or catastrophic releases.
- Backfilling and sealing of the mine workings during mine reclamation/closure.

- Recycling of process water, smelter slag, and air pollution control dust.
- Monitoring and elimination of discharges to surface water, groundwater, soils, and air.
- Replenishment of surface and ground waters with treated effluents.
- Road closure and reclamation (following recontouring) with revegetation of native species.

Although the reclamation of mined lands is often unsatisfactory for ecological habitat restoration, reforestation with native trees has been demonstrated (Plass 1975) and would serve to reduce the abundance of nest parasitic brown-headed cowbirds and restrict their access to mature forest.

Wetlands

Mitigation of wetlands destruction and degradation is the subject of a growing body of literature (Kusler and Kentula 1989). Restoration and mitigation banking concepts are still being evaluated as effective mitigation measures for direct wetlands alterations.

Guidelines for Reviewers

Reviewers of environmental impact assessments will find this document useful if they follow the steps laid out in the introduction:

1. Review the status and trends of habitats in the region.
2. Identify the habitats of concern.
3. Link the activities involved with impacts to these habitats of concern.
4. Devise appropriate mitigations for the impacts.

Each reviewer can then determine the adequacy of the environmental impact assessment in question and recommend modifications to enhance its effectiveness.

In identifying the habitats of concern, the reviewer should supplement the information in this document with detailed locational information on the abundance and distribution of habitats within the region of interest, and with any historical information on the extent and quality of these habitats. Most important, the reviewer should characterize the habitats in terms of their ecological values (e.g., use of wooded wetlands by migratory waterfowl).

In considering the links between activities and habitats, the reviewer should look beyond direct impacts to indirect and subtle effects, including cumulative impacts, interactive and synergistic impacts, and scale-dependent impacts (e.g., effects of fragmentation on ecosystem integrity and species home ranges).

In devising possible mitigations, the reviewer should follow the seven principles for habitat mitigation repeated below. The reviewer should also determine whether adequate assurances have been given that the mitigations proposed will be completed.

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6. Tailor management to site-specific environmental conditions and to the unique impacts of the specific degrading activity.

7. Monitor for habitat impacts and revise mitigation plans as necessary.

Finally, the reviewer should consider the proposed activities and mitigations in the context of relevant regional program goals and objectives (e.g., whether the outcome of the project will be in accordance with principles set out by regional planning commissions such as those established for the Columbia River Basin).

Contacts and Information Sources

When considering habitat conservation issues in an environmental impact assessment for the *Western Forests*, the reviewer should consult the following organizations and individuals for information on habitat impacts and mitigations:

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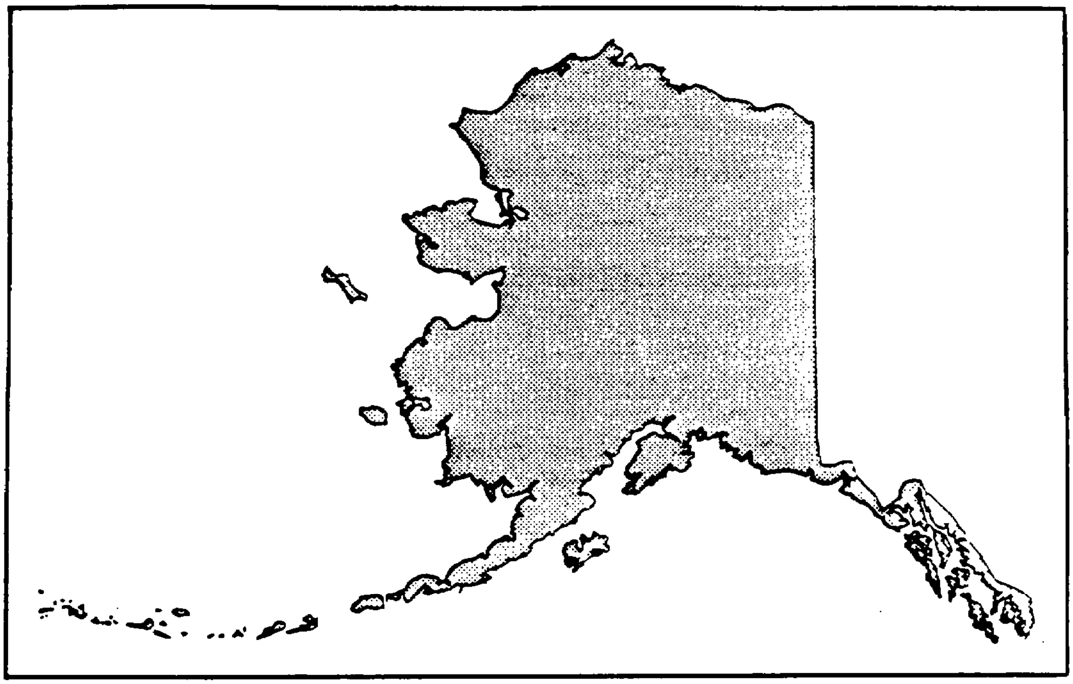
Patrick Bourgeron, Regional Ecologist, The Nature Conservancy
Jim Teeter, Associate Director, Fish and Wildlife Enhancement, U.S. Fish and Wildlife Service

ALASKA HABITAT REGION: *ALASKA*



Geographical Description of the Region

The Alaska Habitat Region consists of the state of *Alaska* and is contained in EPA Region 10. Although only the single state is included, *Alaska* constitutes one-third of the land area of the United States. Also, because it is separated from the conterminous states, *Alaska* contains a unique set of habitat types.

Alaska comprises 5 ecoregions (Bailey 1980). The vegetation of *Alaska* consists of grasses, sedges, lichens with willow shrubs, birch-lichen woodland, needleleaf forest, cottongrass-tussock, dwarf shrubs, lichens, mosses, dwarf birch, Labrador-tea, cinquefoil, white spruce mixed with cottonwood, balsam poplar, willow rose, dogwood, berry bushes, dwarf arctic birch, crowberry, arctic willow, resin birch, dwarf blueberry, cottongrass, bluejoint, taiga, green and thinleaf alder, dogwood, sphagnum, bog rosemary, white mountain-avens, moss-campion, black oxytrope, arctic sandwort, alder thickets, devils club, mountain ash, and alpine-azalea.

Alaska is unique among the regions of the United States in that it still possesses large areas of pristine landscape. The scale of the state is vast, and changes to the landscape from different land use patterns, although increasing, are still primarily restricted to urban centers, fishing ports, and oil and gas producing operations.

Habitats of Concern

Alaska contains many habitats of concern; the most obvious fall into five general categories: old-growth forest, riparian watersheds and fisheries, tundra, maritime forest, and boreal forest. The principal habitats of concern most at risk in *Alaska* are listed below.

PRINCIPAL HABITATS OF CONCERN IN *ALASKA*

1. Old-growth forest of southeastern Alaska
2. Riparian watersheds and salmon rivers
3. Tundra
   - wetlands (e.g., muskeg and sedge meadow)
   - arctic tundra foothills and uplands
   - alpine tundra
4. Aleutian Island maritime grasslands
5. Boreal forest of south-central Alaska

Habitat Values and Trends

The scale and range of habitat types that occur in *Alaska* are unparalleled in the contiguous United States. Large areas of *Alaska* are still without any ground inventories or meaningful ecological descriptions.

Tundra

Alaska contains 173 million ac of rangeland mostly in arctic and alpine tundra. By many definitions, the tundra of *Alaska* is wetland and includes many wetland complexes such as muskeg and sedge meadow. These areas support large populations of caribou, moose, and about 30,000 reindeer. Also present are bears, wolves, coyotes, foxes, squirrels, and mice. Lichen is a primary ground cover in *Alaska*, and it is critical to the survival of reindeer. Lichen habitat has been seriously degraded by overgrazing and wildfires. In the arctic tundra and Bering tundra provinces, cottongrass-tussock is widespread; in the Brooks Range region, lower elevations may be vegetated with sedges and shrubs (USDA Forest Service 1989).

Tundra provides critical habitat for waterfowl; it also supports fisheries on the lowlands and black-tailed deer on the uplands. In the North Slope foothills, caribou use the uplands for calving and are seasonally dependent on tundra vegetation. The tundra and maritime grasslands of the Aleutian system provide one of the outstanding pristine ecosystems in the United States.

Forests

Alaska is less than 40% forested. Today, Alaskan forests consist of 116 million ac of fir-spruce and 11 million ac of hemlock-Sitka spruce (USDA Forest Service 1989). More than 90% of the commercial coastal forests are still in old growth; however, in the interior more than 50% are in young stands (Barrett 1980). Except in the immediate vicinity of villages, the native Indians made no impact on the coastal forests. However, both aboriginal and modern cultures have altered the interior forest through fire.

The mainland of coastal Alaska and the island archipelago contain one of the largest pristine rainforest and shoreline ecosystems in the world. Of this, 11,600,000 ha fall within the Tongass and Chugach National Forests and the Glacier Bay National Park and Preserve. Southeast Alaska is 46% forested, with the remainder in alpine, permanent snow and ice (including broad piedmont glaciers at the northern tip), or bog (muskeg). This coastal forest type (Sitka spruce-western hemlock) extends westward across south-central Alaska where the state is only 11% forested. Similar to the Pacific Northwest, Alaska old-growth forest is multi-aged with codominants 200 to 250 years of age. However, Alaskan old-growth forest experiences less frequent natural perturbations (such as fire) and contains a greater percentage of total closed-canopy cover. Highly productive old-growth forests usually occur in smaller patches than in the Pacific Northwest and are increasingly fragmented toward their northern range limit. In general, however, Alaskan old-growth forest is abundant owing to the relatively low frequency of catastrophic disturbance (Alaback and Juday 1989). Coastal Alaskan old growth supports Sitka black-tailed deer and other wildlife species.

Alaska Coastal Plain

The Alaska Coastal Plain is one of the last intact arctic ecosystems. It supports caribou, musk-ox, moose, Dall sheep, wolf, arctic fox, brown bear, and 22% of the western arctic population of lesser snow goose. This area is threatened by oil and minerals exploration and development; in many cases land is being leased to oil companies by native corporations (Frazier 1987). Oil drilling in Prudhoe Bay has caused erosion, vehicle damage, heavy dust loads from the road system, and water damming and tundra ponding.

Aquatic Systems

About 16 million ac of *Alaska* is in water area, principally the coastal waterways, the numerous large rivers of the Yukon system, and more than 3 million lakes more than 20 ac in size. *Alaska* possesses the world's most productive salmon fisheries.

Activities and Impacts Affecting Habitats

The following activities result in the major impacts on habitats of concern in *Alaska*.

IMPACTS ON HABITATS OF CONCERN IN *ALASKA*				
	Land Conversion	Timbering	Mining	Other
Old-growth forest	Minor	Heavy logging in the Southeast	Minor	Minor
Riparian watersheds and fisheries	Urban development in river bottoms and development of pipeline and transportation corridors	Major impact from logging practices and sedimentation	Degradation from in-stream placer mining	Minor
Tundra	Conversion around urban centers	None	Impacts of oil and gas production	Impacts of military activities
Maritime grasslands	Minor	None	Minor	Impacts of military activities
Boreal forests	Minor	Moderate	Minor	Minor

Land Conversion

Alaska is experiencing rapid development of certain areas, especially around Anchorage and Fairbanks (Mary Lynn Nation, personal communication). This includes urban sprawl and the building of infrastructure for tourism. Considerable conflicts with wetland fills have arisen because of the extent of tundra wetland. Land conversions include areas for ports and airports infrastructure, and areas for harbors and the shipping industry. Private fish hatcheries and ladders are consuming land in the south-central region, and the fishing industry in Dutch Harbor has converted land for processing and storage operations. One of the greatest threats is posed by transportation corridors; a recent proposal is to open the Dalton Highway (the hauling road to the North Slope) to recreation.

Timber Harvesting

Timbering of Alaska is principally confined to the coastal southeastern area of productive Sitka spruce-hemlock. It ranks with tourism, behind oil production and fisheries, as the state's major industries (USDA Forest Service 1989). Considerable research has been conducted on timbering methods for this area and will likely result in both less national forest area being available for logging and more intensive timbering of the remaining lands. The increase in privately owned forest will likely result in logging and a decreased forest area in certain locations. In particular, the leasing of land through native corporations has resulted in increased logging.

Timbering activities include clear cuts and conversions for roads, antennas, and other operational areas. Severe impacts are also caused by log transfer, staging, and in-water storage. Negative effects include erosion and siltation of salmon fishery habitat and loss of habitat for black-tailed deer.

Mining

In addition to timbering, mining and oil and gas development pose severe risks to the pristine natural areas of *Alaska*. Exploration and production of oil reserves in *Alaska* are in direct conflict with many wildlife requirements. In addition to the production on the Kenai Peninsula oil patches and offshore oil drilling in Cook Inlet, considerable small-scale drilling exploration is conducted in undeveloped areas. Discovery of oil in these regions would require substantial infrastructure development, including pipelines and tankering. New petroleum and liquid natural gas (LNG) pipelines are also proposed.

Gold mining is another cause of habitat degradation in *Alaska*. This includes placer mining and proposed copper leachate facilities. Impacts include the effects of tailings and runoff, especially the contribution to erosion and sedimentation that negatively affect salmon fisheries.

Military Activities

Military operations constitute another activity degrading habitats in *Alaska*. This is most important in the pristine Aleutian maritime grasslands; the fact that these areas are generally inaccessible has prevented virtually all other degradation. Impacts include toxic releases and bulldozing operations causing erosion.

Mitigations of Impacts

The conservation of habitats requires consideration of mitigations for the major activities impacting habitats of concern. In *Alaska*, the primary habitat impacts are caused by the following:

- Timbering of old-growth forests in southeastern Alaska.
- Mining impacts on tundra and aquatic systems.
- Urban expansion and conversion of tundra environments.
- Impacts of logging and development on riparian areas and salmon fisheries.

Management of the combined effect of these activities on sensitive habitats requires a holistic, ecosystem-level approach. In particular, the approach pays special attention to the needs of wide-ranging species such as caribou. It emphasizes the need to look at the landscape scale (not institutional boundaries) for the implications of habitat value and modification.

Timber Harvesting

At a minimum, the production of commercial wood products from an area must not exceed the sustainable level if the ecological integrity of a forested area is to be maintained. Where sensitive forest types exist, logging may be completely prohibited or constrained to specific methods to prevent habitat loss or degradation. In other areas, more extreme harvesting methods may be allowed or prescribed to establish or maintain desired forest conditions. Acceptable methods will vary according to local forest ecology and the desired future condition of the site. Analysis of harvesting techniques must be based upon an analysis of the structure and diversity of the forest canopy, midstory, and understory.

A recent directive of the Chief of the U.S. Forest Service acknowledges this fact and points out that clear cutting is acceptable only when needed to replicate natural ecological processes. Selective cutting can preserve forest structural diversity, the primary determinant of wildlife habitat (Harris et al. 1979). However, it can reduce horizontal diversity (NRC 1982). The harvesting technique employed must be based upon sound silvicultural prescriptions and demonstrate its capability to maintain vertical diversity (foliage height diversity), horizontal diversity (interspersion, edge, juxtaposition, patchiness), and a mixture of live and dead wood. Specific timber harvesting operations should be designed to preserve the structure and diversity of the natural forest habitat.

An important component of selective cutting should be the preservation of standing dead trees. Many birds nest, roost, or forage for invertebrates in standing trees with decayed wood. These cull trees are usually the first focus of forest-thinning operations, to the detriment of the birds. Breeding bird abundance declines rapidly following a clear cut, and the species composition continues to change for 10 to 15 years (DeGraaf 1991). However, if trees with cavities are saved, many of these species can successfully forage on sound boles. About one large cavity or den tree per 2 ha is required for population of large species such as wood ducks; this requires harvest rotations of 100 to 125 years (although rotations of 65 years produce trees large enough for species nesting in smaller cavities).

Responding to the "biodiversity crisis," the U.S. Forest Service is moving toward an ecosystem approach to forest management (Bob Szaro, personal communication). Recent forest management plans have incorporated tenets of the "New Forestry" espoused by Jerry Franklin. These progressive plans require the rigorous implementation of ecological management practices to maintain forest productivity and to preserve the functioning of sensitive forest components such as old-growth or late-successional forests. Effective mitigations for habitat conservation in forest management require specific management measures at the site, watershed, and landscape levels. For example, the location and size of timber harvests should be planned to minimize reduction of the core area of mature forest (e.g., harvest only alternate basins until regrowth). Maintenance of mature-forest stands in managed landscape can be achieved by extending rotation (beyond 80) to 150 to 200 years, by leaving some stands unharvested for old growth, and by linking stands. Landscape-scale considerations include the provision of buffer zones and habitat corridors as were discussed in the introduction of this document. Management measures recommended for conserving habitat within managed forests include the following:

- Minimize the construction of new roads and close roads not in use either permanently or seasonally.
- Use best management practices (BMPs) such as filter strips to minimize erosion during harvesting or road construction.
- Maintain 100-ft riparian zones with adjacent feathered transition zones to buffer edge effects.
- Restrict harvesting operations to periods when the ground is either dry or frozen.

- Maintain site productivity by retaining large woody material and minimizing mineral soil exposure and compaction during harvesting.

- Manage for natural disturbance patterns to maintain natural openings and successional-stage composition.

- Maintain connections between blocks of interior forest, especially old growth.

- Provide for the protection of special areas, including cliffs, caves, taluses, riparian areas, and old-growth stands.

- Maintain the structural integrity and the native variety of the forest by managing for the natural composition of the following components: vegetative types, seral stages, tree types and sizes, standing dead trees and down material, tree snags, and cavity trees.

The conservation of old-growth forest presents a special challenge that is currently being addressed in *Alaska*. In southeastern Alaska, the rainforest extends 500 miles long by 100 miles wide across a mosaic of offshore islands. The forest supports Sitka spruce 200 feet tall and 400 years old with a lush undergrowth of evergreen plants, ferns, and mosses. Most of this rainforest is within the confines of the Tongass National Forest and is subject to the multiuse management and timber harvesting of the U.S. Forest Service. Forest series with late successional components in the Tongass include upland, riparian, and beach Sitka spruce, Sitka spruce-western hemlock, mixed conifer, and subalpine mountain hemlock. An old-growth management prescription for the Tongass prepared by a recent workgroup (Samson et al. 1991) included the following requirements: (1) define ecological units; (2) establish a province system that captures representative habitat for dependent species; and (3) recommend the size, shape, and distribution of habitats to maintain viable populations of species. The group recommends that at least one watershed within each province be left intact for wildlife. Timber and timber-wildlife emphasis alternatives were described. The latter requires that forest management in the Tongass include the following:

- Harvest areas from the periphery inward to maintain large continuous blocks.

- Harvest areas so that they are "sloppy" with small patches of green trees, brushy openings, and snags to increase the habitat available through time.

- Provide edges that are "feathered" to reduce vulnerability to windthrow.

- Harvest habitat types in a manner that ensures the continued existence of each type and relative availability of each type.

- Use habitat models for indicator species to prioritize areas to be retained as old-growth wildlife habitat.

Based on population models of ermine, islands of less than 2,000 ac of forest habitat should not be logged. Alternatively, clusters of smaller islands may withstand timbering if species have appropriate dispersal routes.

Mining

Mitigation of mining impacts involves siting issues, technological solutions to eliminate contamination, and restoration programs. The major mitigations for oil and gas extraction and production are the proper sitings of rigs, reserve pits, processing facilities, and roads where they will have minimal impacts on habitats of concern. Most important for coal and mineral mining is the siting of mining operations and tailing ponds to avoid habitats of concern, wetlands, riparian areas, and recharge areas. Specific mitigation measures depend on the type of mining and the specific process causing impacts. It is generally best to minimize the area affected as it is unlikely that even the disrupted soils and sediments can be restored. In addition to minimizing the area disturbed, activities should be timed to avoid disturbing nearby plants and animals during crucial periods of their life cycle.

Possible mitigation measures for mining operations are listed below (SAIC 1991a, 1991b):

- Design of mine entrances and workings to minimize future mine drainage.
- Runon and runoff control measures such as berms and ditches.
- Adequate depth and lining of pits for containment of muds and leachate.
- Elimination of migration of fluids through casings and dewatering.
- Separation of wastes and contaminated soils with proper disposal.
- Treatment of leach heaps and neutral or acidic wastewaters to reduce the load of cyanide, nitrates, and heavy metals.
- Closure planning that addresses hydrology, geochemical controls, treatment, and restoration.
- Nets or other covers over process ponds.
- Maintenance of an anaerobic environment in the tailing pile during periods of inactivity.
- Secondary containment of tanks and contingency plans for sudden or catastrophic releases.
- Backfilling and sealing of the mine workings during mine reclamation/closure.
- Recycling of process water, smelter slag, and air pollution control dust.
- Monitoring and elimination of discharges to surface water, groundwater, soils, and air.
- Replenishment of surface and ground waters with treated effluents.
- Road closure and reclamation (following recontouring) with revegetation of native species.

Although the reclamation of mined lands is often unsatisfactory for ecological habitat restoration, reforestation with native trees has been demonstrated (Plass 1975) and would serve to reduce the abundance of edge species and restrict their access to mature forest.

Land Conversion

Effective mitigation of land conversion activities can sometimes be obtained only by avoiding impacts on rare or unusual habitat types. Rarely, if ever, is restoration or compensation an adequate mitigation for the loss of these habitats. In these cases, mitigation is a siting issue, where construction and degrading activities are located at a distance from the habitats of concern. The habitat is adequately preserved if all possible impact scenarios are accounted for. Barring this solution, effective management measures must be implemented to ensure the protection of the habitats of concern.

In the case of unique tundra habitats, hydrological and contamination concerns are especially important. Construction or resource management activities must take special precautions to minimize mechanical disturbance of permafrost soils. Road building and structural "improvements" must not result in altered hydrological regimes. Where rare plant types exist or where habitats are unstable, recreational access may have to be limited. These mitigations can be best implemented by creation of a regional land-use plan (through a coordinating council like the Waterfowl Flyway Council) and landowner incentives like the Conservation Reserve Program.

Wetlands

Mitigation of wetlands destruction and degradation is the subject of a growing body of literature (Kusler and Kentula 1989). Restoration and mitigation banking concepts are still being evaluated as effective mitigation measures for direct wetlands alterations.

Guidelines for Reviewers

Reviewers of environmental impact assessments will find this document useful if they follow the steps laid out in the introduction:

1. Review the status and trends of habitats in the region.

2. Identify the habitats of concern.

3. Link the activities involved with impacts to these habitats of concern.

4. Devise appropriate mitigations for the impacts.

Each reviewer can then determine the adequacy of the environmental impact assessment in question and recommend modifications to enhance its effectiveness.

In identifying the habitats of concern, the reviewer should supplement the information in this document with detailed locational information on the abundance and distribution of habitats within the region of interest, and with any historical information on the extent and quality of these habitats. Most important, the reviewer should characterize the habitats in terms of their ecological values (e.g., use of wooded wetlands by migratory waterfowl).

In considering the links between activities and habitats, the reviewer should look beyond direct impacts to indirect and subtle effects, including cumulative impacts, interactive and synergistic impacts, and scale-dependent impacts (e.g., effects of fragmentation on ecosystem integrity and species home ranges).

In devising possible mitigations, the reviewer should follow the seven principles for habitat mitigation repeated below. The reviewer also should determine whether adequate assurances have been given that the mitigations proposed will be completed.

1. Base mitigation goals and objectives on a landscape-scale analysis that considers the needs of the region.

2. Mimic natural processes and promote native species.

3. Protect rare and ecologically important species and communities.

4. Minimize fragmentation of habitat and promote connectivity of natural areas.

5. Maintain structural diversity of habitats and, where appropriate, species diversity to promote the natural variety of the area.

6. Tailor management to site-specific environmental conditions and to the unique impacts of the specific degrading activity.

7. Monitor for habitat impacts and revise mitigation plans as necessary.

Finally, the reviewer should consider the proposed activities and mitigations in the context of relevant regional program goals and objectives (e.g., whether the outcome of the project will be in accordance with principles set out by regional planning commissions).

Contacts and Information Sources

When considering habitat conservation issues in an environmental impact assessment for *Alaska*, the reviewer should consult the following organizations and individuals for information on habitat impacts and mitigations:

State Natural Heritage Programs
U.S. Fish and Wildlife Service, Regional and Area Offices
State Fish and Game Departments
University and Research Programs
Herbaria and Museums

Gerry Tande, Regional Ecologist, The Nature Conservancy
Mary Lynn Nation, Division of Federal Activities, U.S Fish and Wildlife Service, Alaska

HAWAII AND THE ISLAND TERRITORIES

This section briefly discusses *Hawaii and the Island Territories*. They comprise a relatively small land area, but are sufficiently distinct to require discussion separate from the seven major regions of the United States.

Geographical Description

The Hawaiian Islands and the Pacific Trust Territories of Guam and the Northern Marianas are all included in EPA Region 9. Puerto Rico and the U.S. Virgin Islands are included in EPA Region 2. Hawaii forms its own ecoregion in the Bailey system (1980), a Highland Ecoregion within the Rainforest Division. The island territories have not been classified into ecoregions by either the Bailey or Omernik systems.

The vegetation of the Hawaiian Islands comprises tropical shrubs, dense needleleaf and broadleaf forests, bogs, and moss lichen communities. Because of its isolation, Hawaii contains many endemic species and possesses a fauna and flora unlike that found anywhere else. Although the community compositions are different, the vegetation of the Pacific Trust Islands and Puerto Rico and the Virgin Islands contains many of the same elements of tropical forests and island floras.

Habitats of Concern

The diverse ecosystems of Hawaii can be classified as existing on dry leeward or wet windward areas. Leeward lowlands consist mostly of introduced plants such as kiawe and haole koa in grassland or savanna habitats. Leeward uplands contain evergreen scrublands and forests with exotics such as guava, Java plum, and Christmasberry. Windward evergreen rainforests are dominated by native ohia and koa, and constitute 0.3 of the 1.7 million ac of forest on Hawaii. Above the rainforest on the highest islands of Maui and Hawaii are zones of mountain parklands of koa and mamane, alpine scrub, and alpine tundra (USDA Forest Service 1989).

Of the 150 vegetation types in the Hawaii Natural Heritage classification, more than 50% are rare and nearly all are endemic (Sam Gon, personal communication). Principal habitats of concern include the following:

- Brackish anchialine pools along the shore.
- Coastal and lowland wetlands (below 3000 ft).
- Coastal and lowland forest and shrub ecosystems.
- Upland forest types.
- Wet bog ecosystems within forests.
- Subalpine and alpine zones.

Habitat Values and Trends

All the major ecological zones are represented in the 6,500 sq mi of Hawaiian land mass. More than 10,000 species of plants and animals are endemic to Hawaii. Extinctions of native species began with the arrival of Polynesians 1,500 years ago and accelerated with the arrival of Europeans in the late

1700s, reaching rates thousands of times the natural rate. Of the 140 bird species native to Hawaii, 70 have become extinct and 30 more are endangered. Currently, 37 species of plants in Hawaii are federally listed as endangered and 152 more are expected to be listed in the next 2 years (Hawaii State Department of Land and Natural Resources et al. 1991).

The aboriginal Hawaiians converted most of the land below the 600-meter elevation to agriculture on the eight main islands. Today, nearly two-thirds of Hawaii's original forest cover and 50% of the rainforest have been lost to land conversion for housing, agriculture, and ranching. Ninety percent of the lowland plains dry forests, 61% of the mesic forests, and 42% of the wet forests have been destroyed. The last remnants of Hawaiian coastal plant communities are on the most remote and arid shores. The unique terrestrial environments of Hawaii are also being degraded or lost due to the logging of tropical forests. Hawaii contains 180 terrestrial ecosystems, of which at least 88 ecosystems will be lost within 20 years unless current losses of habitat are addressed (Tangley 1988). Similar histories have befallen the Pacific Trust Territories of Guam and the Northern Marianas and Puerto Rico and the U.S. Virgin Islands. For example, the loss of tropical rainforest to timbering and conversion to agriculture is a major problem in Puerto Rico.

The invasion of non-native species represents the greatest threat to surviving native species and natural communities on all the U.S. islands. The Hawaiian archipelago has lost more than 75% of its original endemic land bird fauna through prehistoric and historic extinctions; the comparable Galapagos archipelago as a whole is not known to have lost a single land bird species (Loope et al. 1988). The absence of native large mammals has left the native fauna and flora vulnerable to the browsing, rooting, and trampling of introduced pigs, goats, cattle, and deer. On Hawaii's 1.4 million ac of rangeland, most native plants have been replaced by introduced perennials. Native Hawaiian birds have suffered from avian malaria spread by introduced mosquitos, and native plants have been smothered by the exotic banana poka. In Guam, the introduced brown tree snake has wiped out 9 of the 11 species of native birds, and Hawaii is now threatened by the repeated reintroduction of this reptile.

Activities and Impacts Affecting Habitats

The majority of forest land remaining in Hawaii is contained within the state forest reserves and conservation districts. These lands are managed principally for watershed and aquifer protection and allow little commercial wood harvesting. However, timbering of native koa and exotic eucalyptus do occur, and logging continues to impact private lands.

A greater threat to forest ecosystems in Hawaii is livestock grazing. Substantial areas of forest continue to be cleared to promote forage growth for cattle ranching (USDA Forest Service 1989). The current tax structure in Hawaii encourages clearing of forest for ranching.

Conversion of lands for urban and resort construction has a major impact on coastal and lowland environments. In addition, growing commercial and residential development contributes to the loss of dry areas subject to fire. This problem is exacerbated on military firing ranges.

Agriculture has long been an important industry on Hawaii, and it continues to impact adjacent terrestrial and aquatic habitats through sedimentation and contamination with pesticides.

The primary threat to *Hawaii and the Island Territories* is alien species. Introduced species contributing to habitat destruction include herbivorous mammals, predaceous ants, dogs, cats, mongoose, alien arthropods, mollusks, and alien plants. Wet ecosystems, in particular, are threatened by invading non-native animals (principally pigs, goats, deer) that disrupt the natural vegetation to the extent that native species are replaced by non-native plants. The invasion of combustible non-native weeds has created a cycle of wildfires that often destroy rare dryland native plants (Hawaii State Department of Land and Natural Resources et al. 1991).

The decline and extinction of many endemic Hawaiian bird species can be attributed to the unprecedented invasion of exotic species. Among exotic birds, more introductions (162) and establishments (between 45 and 67) have occurred in Hawaii than anywhere else in the world (Scott et al. 1986). Today, more than 80 introduced vascular plant species currently pose threats to the native biota in Hawaii. The inadequacy of detection and control has resulted in continuing invasions, and the problem of existing exotics requires constant management or additional losses will result. It is believed that biological methods offer the best hope of extensive long-term control of the most aggressive alien plants in natural systems.

Guidelines for Reviewers

Reviewers of environmental impact assessments for *Hawaii and the Island Territories* should refer to other regional discussions for more detailed information on habitat impacts and their mitigations. The following section outlines the consideration of habitat conservation in the review process:

1. Review the status and trends of habitats in the region.

2. Identify the habitats of concern.

3. Link the activities involved with impacts to these habitats of concern.

4. Devise appropriate mitigations for the impacts.

Each reviewer can then determine the adequacy of the environmental impact assessment in question and recommend modifications to enhance its effectiveness.

In identifying the habitats of concern, the reviewer should supplement the information in this document with detailed locational information on the abundance and distribution of habitats within the region of interest, and with any historical information on the extent and quality of these habitats. Most important, the reviewer should characterize the habitats in terms of their ecological values (e.g., use of wooded wetlands by migratory waterfowl).

In considering the links between activities and habitats, the reviewer should look beyond direct impacts to indirect and subtle effects, including cumulative impacts, interactive and synergistic impacts, and scale-dependent impacts (e.g., effects of fragmentation on ecosystem integrity and species home ranges).

In devising possible mitigations, the reviewer should follow the seven principles for habitat mitigation repeated below. The reviewer should also determine whether adequate assurances have been given that the mitigations proposed will be completed.

1. Base mitigation goals and objectives on a landscape-scale analysis that considers the needs of the region.

2. Mimic natural processes and promote native species.

3. Protect rare and ecologically important species and communities.

4. Minimize fragmentation of habitat and promote connectivity of natural areas.

5. Maintain structural diversity of habitats and, where appropriate, species diversity to promote the natural variety of the area.

6. Tailor management to site-specific environmental conditions and to the unique impacts of the specific degrading activity.

7. Monitor for habitat impacts and revise mitigation plans as necessary.

Finally, the reviewer should consider the proposed activities and mitigations in the context of relevant regional program goals and objectives (e.g., whether the outcome of the project will be in accordance with principles set out by regional planning commissions).

Contacts and Information Sources

When considering habitat conservation issues in an environmental impact assessment for the *Hawaii and the Island Territories*, the reviewer should consult the following organizations and individuals for information on habitat impacts and mitigations:

State Natural Heritage Programs
U.S. Fish and Wildlife Service, Regional and Area Offices
State Fish and Game Departments
University and Research Programs
Herbaria and Museums

Sam Gon, Regional Ecologist, The Nature Conservancy
Jim Teeter, Associate Director, Fish and Wildlife Enhancement, U.S. Fish and Wildlife Service

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Biological and Social Issues Related to Confinement of Wild Ungulates

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Foreword

Presidents of The Wildlife Society occasionally appoint ad hoc committees to study and report on selected conservation issues. The reports ordinarily appear in 2 related series called either Technical Review (formerly "White Paper") or Position Statement. Review papers present technical information and the views of the appointed committee members, but not necessarily the views of their employers. Position statements are based on the review papers, and the preliminary versions ordinarily are published in *The Wildlifer* for comment by Society members. Following the comment period, revision, and Council's approval, the statements are published as official positions of The Wildlife Society.

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SYNOPSIS

Commercial demand for hunting and for sale of live ungulates and their products has prompted the growth of a commercial industry that raises non-domesticated native ungulates within managed properties. These properties vary in size, management intensity, and product, but typically include fencing designed to control animal movements. Animals often are confined to fulfill management goals related to population management for harvest and commercial production of live animals or their products (e.g., venison, velvet, semen). The rapid expansion in number and acreage of fenced properties throughout North America has prompted a need for a review of the biological and social issues related to these management practices. In this report we review the primary biological and social issues directly and indirectly associated with confinement of wild ungulates, defined as all hoofed mammals.

Fencing is used to control movement of animals to improve population-level management effectiveness and for commercial production of live animals or their products. Fences used to confine ungulates have various names representing either the type of fence or the reason for confinement, such as high fence or game fence. The most typical high fence consists of 2.4-m-high net wire. The extensive use of high fences began in Texas in the 1930s, but has expanded to other states and provinces in recent years. For example, in Wisconsin there are 947 high-fenced facilities containing 35,000 captive cervids—deer and elk.

Biological issues related to confined ungulates include behavioral impacts on target species, diseases associated with confinement, genetic impacts of confinement and shipment across natural ranges, habitat impacts, and impacts on non-target species. The goal of high-fence construction is to modify animal movement patterns. While most animals adapt to such a change, exclusion from critical or migratory habitats may impact survival or production. The migration patterns of free-ranging animals may be disrupted if high fences surround critical migratory range or block migratory corridors. Fences can reduce egress of animals and facilitate control of ungulate density, sex ratio, and age structure to improve local management effectiveness. Some fenced populations do not receive harvest rates sufficient to control population density, resulting in overpopulation. Infectious diseases are a concern when transmission is increased at high densities, when host animals are subjected to nutritional or environmental stressors, and when animals of different species and sources are mixed. The risk of disease transmission between captive and free-ranging animals depends upon the management circumstances (e.g., fencing, geography, etc.), the likelihood of direct or indirect contact between the captive and free-ranging species, and the routes of transmission of any given pathogen. Chronic wasting disease (CWD), bovine tuberculosis (TB), and meningeal worm are of most concern to wildlife managers. Control or eradication of CWD is extremely difficult given the long incubation period, absence of practical antemortem diagnostic tests, an extremely resistant infectious agent possibly leading to environmental contamination, and limited knowledge of the mode of transmission. Management currently involves quarantine or depopulation of captive CWD-affected herds, significant population reduction of wild populations, and banning translocation and artificial feeding of cervids in the endemic areas. In the absence of complete information, the public health concerns about CWD remain important. Bovine tuberculosis in farmed cervids has been a serious problem in North America since the 1980s and has been the subject of a state–federal eradication program. The genetic impacts of enclosed populations and the mixing of genetically distinct populations is unclear. If escapees breed with free-ranging natives, there could be dilution of unique genetic stocks and reduced fitness. The potential impact of escapees would be proportional to the number, survival, and reproductive success of escaping animals and the severity of selective disadvantage for any maladaptive traits. The long-term genetic effects of intensive management strategies within enclosures are unknown. Vegetative diversity and ecological health decline when ungulate populations are allowed to exceed habitat carrying capacity. However, ecological health within fenced habitats may increase when improved effectiveness of population control is combined with habitat management and regulation of livestock grazing pressure.

Social issues related to confined ungulates include ownership of wildlife resources, hunter ethics, the public perception of hunting, commercialization and domestication of wild animals, and ecological stewardship. The North American system of wildlife management is based on the premise that endemic wildlife belongs not to individuals but to the people of the state, and responsibility for managing that wildlife is entrusted to the governmental regulatory agency. The application of the "public trust doctrine" to wildlife is deeply rooted in history, beliefs, and court opinions. Defining sportsmanship and describing a satisfactory hunter ethic for modern conditions have been controversial. A "canned" hunt describes a situation in which the client pays to kill a specific type of animal under conditions where the probability of failure is greatly reduced. We consider this type of practice unethical. This activity could be used by anti-hunters in their attempts to sway public opinion against hunting. Efforts are needed to improve the management, practice, and image of hunting. A major impetus for expansion of the game farm and hunting industries in North America has been agricultural diversification. There are 4 primary products in the game farm industry: meat, velvet, breeding stock, and shooter bulls. The venison and velvet market niche is limited and mostly filled by foreign producers. The market for breeding stock has been impacted by a United States Department of Agriculture declaration of CWD as an animal health emergency and state actions related to CWD. Wildlife managers recently have begun to discuss the ethics of wildlife management as practiced in the modern world. Professional wildlife managers should encourage ecological stewardship as the basis for management actions.

INTRODUCTION

Commercial demand for hunting and for sale of live ungulates and their products has prompted an expansion of a commercial industry that raises non-domesticated native ungulates within managed properties. These properties vary in size, management intensity, and product, but typically include fencing designed to control animal movements. The rapid expansion in number and acreage of fenced properties throughout North America and the activities associated with these facilities has generated a variety of biological and social issues at state, national, and international levels.

Fencing as a management tool has been described as existing along a "corral continuum" (Stedman 1998). On the least intensive end of this continuum are properties where the fencing simply encloses large areas of natural habitat with the objective of improving effectiveness of "traditional" population management, such as manipulation of density, sex ratio, and age structure. On the most intensive end of the continuum are properties where fencing is used to manipulate genetic composition within small breeding pens using controlled breeding or artificial insemination. Brood stock must be obtained, which requires private ownership, sale, and shipment of animals among breeding facilities. To facilitate record keeping, animals are clearly marked using livestock ear tags or freeze branding. At this most intensive end, pens are too small and animal density is too high to allow natural provision of habitat requirements, so husbandry must fulfill nutritional requirements by providing full-ration feed and water. Infectious diseases are a concern whenever animals are maintained at high densities, and shipment of diseased or exposed animals among facilities is problematic.

Our primary objective is to review the important biological and social issues associated directly and indirectly with confinement of non-domesticated, native ungulates. Biological issues include behavioral impacts on target species, diseases associated with confinement, genetic impacts of confinement and shipment across natural ranges, habitat alteration, and unintended effects on non-target species. Social issues include ownership of wildlife resources, recreational ethics related to fair chase and "canned hunts," the public perception of hunting, commercialization and domestication of wild animals, and ecological stewardship within the wildlife profession.

Our secondary objective is to include a discussion of the implications of non-domesticated, native ungulate confinement on our natural resources.

OVERVIEW OF FENCING

High fences may be used in conjunction with intensive ungulate management to prevent egress of animals and to increase effectiveness of actions to manipulate density, sex ratio, and age structure by limiting animal movement between properties. High fences control access to older-aged males, some of which have been afforded protection from earlier harvest. If harvest rates are adequate, increased control results in ungulate densities lower than surrounding properties. However, some fenced populations do not receive harvest rates sufficient to control population density, resulting in overpopulation. Ungulates may also be confined for commercial production of venison, hide, velvet, hard antler, or breeding purposes. In some jurisdictions, specialized fencing requirements are used to minimize contact between confined and free-ranging populations.

The acreage included within high fenced enclosures varies dramatically. Enclosure size varies from one-acre breeding pens up to population-level management enclosures of 30,000 to 40,000 acres. Generally, the smaller holdings generate the most significant biological and social issues.

In a 2001 survey, 58 American states and Canadian provinces documented the extent and circumstances associated with construction of ungulate enclosures (K. M. Hunt, Department of Wildlife and Fisheries, Mississippi State University, unpublished data). Fencing was allowed in 49 of 58 (85%) responding states–provinces, but 27 of those had some restrictions relating to the practice. Nine of 58 (15%) states–provinces forbade enclosures through a law or agency regulation. Of those forbidding enclosures, 3 states–provinces "grandfathered" enclosures remaining from the period before regulation. Fencing of free-ranging ungulates was not allowed in 43 of 58 (74%) responding states–provinces. Of the 15 (26%) states–provinces that allowed enclosure of free-ranging ungulates, 11 (71%) considered the enclosed animals to be public property even after being enclosed. In 3 of the 4 states–provinces where animals became private property, the enclosure owners were required to pay compensation to the state or province for the enclosed animals. A detailed list of 2002 state regulations related to chronic wasting disease are available from the Michigan Department of Natural Resources at http://www.schmitts@michigan.gov (S. M. Schmidt, Michigan Department of Natural Resources, personal communication).

Physical Characteristics

Fences used to confine ungulates have various names representing either the type of fence or the reason for the confinement. A partial list includes high fence, game fence, elk (proof) fence, and deer (proof) fence.

The most typical high fence is 2.4-m-tall, 12-gauge or greater woven wire fence with vertical stays placed at 15-cm intervals and 17 evenly spaced horizontal wires creating a 15-cm × 15-cm mesh. The woven wire often attaches to and is suspended from a heavier gauge, high tensile wire running between posts. The most common posts are round, pressure-treated wood, 15 cm in diameter or greater, and, depending upon soil conditions, buried 1–2 m deep. Pipe greater than 2.5 cm in diameter may also be used as posts. Occasionally, steel posts are alternated with wooden posts. Corners and stress points are braced. Gates are designed with similar height and mesh characteristics to the fence. Fences may be buried to prevent movement of predators under the fence.

An available variation of the woven wire fence is to stagger the mesh diameter. The spacing of horizontal wires is variable and creates a fence with the lower 1.25 m composed of, in ascending order, 8 × 18-cm, 10 × 18-cm, and 13 × 18-cm mesh sizes. Smaller mesh sizes lower on the fence exclude predators and confine neonates. Fences confining valuable white-tailed deer are frequently extended to 3 m by adding 2 smooth, high tensile wires above the 2.4-m woven wire, with the added benefit of protecting the fence from falling vegetation.

Other fence designs are becoming more common (Palmer et al. 1985). Electric fencing is often attractive because maintenance costs are lower. High tensile wire fences with 5–9 horizontal wires are used in some locales. Low fences may be used to confine non-jumping ungulates such as pronghorn and blackbuck antelope.

Double fencing (i.e., 2 parallel 2.4-m or higher fences situated 2–3 m apart) is required by some regulatory agencies when confining ungulates in situations where escape or direct contact with wild animals would pose significant adverse consequences.

Distribution

The extensive use of high fences to restrict ungulate movements began in Texas in the 1930s as a way to confine exotic wildlife (J. Cooke, Texas Parks and Wildlife Department, personal communication). Under Texas law, exotics are considered livestock and fencing limits their movement onto neighboring properties. The number of exotic species and individuals confined within high fences in Texas has increased steadily; the most recent survey showed that high fences confined 118,000 individuals from 71 exotic species (Traweek 1995). The exotic species most commonly confined in high fences were axis deer, blackbuck antelope, and fallow deer. Roughly 1,000 properties in Texas are high fenced, with the total area confined by fence estimated to be 1.6 million ha, confining about 200,000 whitetails or 4.6% of the statewide white-tailed deer population (E. L. Young, Texas Parks and Wildlife Department, unpublished data). Some fenced areas exceed 8,000 ha (S. J. Williamson, Wildlife Management Institute, personal observation).

Fencing of ungulates has increased greatly in recent years. Mississippi contains a minimum of 65 high-fenced enclosures and most were constructed within the last 10 years; small, private breeding pens are not allowed (L. E. Castle, Mississippi Department of Wildlife, Fisheries, and Parks, personal communication). In Wisconsin there are 947 high-fenced facilities containing 35,000 captive cervids. Of these fenced facilities, 575 contain 17,500 white-tailed deer, 272 contain 10,815 elk, 43 contain 4,480 red deer, 17 contain 149 reindeer, and 40 contain 2,333 exotics such as fallow deer (T. Hauge, Wisconsin Department of Natural Resources, personal communication). From 1994 to 1998 in Michigan, the numbers of captive deer and elk have grown 50% and 100%, respectively, with 21,000 deer and 2,600 elk enclosed in 1998 (Coon et al. 2002). Most enclosures in Michigan are relatively small, with 76% less than or equal to 8 ha (20 ac, Coon et al. 2002).

BIOLOGICAL ISSUES

Behavioral Impacts

<u>Movements and Home Range</u>

When a wild ungulate is confined by the construction of a high fence, movement patterns within the animal's daily or seasonal home range may be altered. Construction of a high fence may also exclude wild ungulates from important habitats located within the fence. While most animals will quickly adapt to such a change, exclusion from critical habitat types may impact survival or production.

In some areas, periodic droughts cause ungulates to wander widely in search of forage or water supplies (Urness 1981, Cooke 1993). Movements would be limited during drought conditions in South Texas today due to the proliferation of high fences (J. Cooke, Texas Parks and Wildlife Department, personal communication). In the northern United States and Canada, periodic severe winter weather causes ungulates to move to winter ranges where forage is most readily available, or where vegetation buffers snow depth. Winter ranges are not static (Kelsall 1969, Nelson and Mech 1986) and weather patterns determine which ranges are chosen within a season, or between years. High fences likely limit the ability of individuals to seek alternate home ranges in times of climatic stress.

Ungulate home range size varies with population dynamics, environmental factors, and habitat quality. Wild ungulates home ranges are frequently extensive (e.g., Demarais et al. 2000), so that only the largest high-fenced areas could possibly provide all components of a typical home range. In

Michigan, for example, 76% of ungulate enclosures were <8.1 ha in size (Coon et al. 2000). Most high-fence operations, therefore, must provide supplemental food and/or water for the confined animals. The negative consequences associated with supplemental feeding, including disease transmission, ungulate overabundance, societal disapproval, and threats to human health and safety, are highlighted in Williamson (2000).

Dispersal and Migration

Ungulate population dynamics regularly include dispersal by young animals. Where habitat quality is patchy, dispersal distances may be significantly greater than norms presented here. Yearling male white-tailed deer typically disperse from 3 to 10 km from natal ranges (Demarais et al. 2000). Black-tailed deer move 12–15 km (Bunnell and Harestad 1983). Elk may move hundreds of kilometers when dispersing (Adams 1982). Moose calves distanced themselves from their mothers by an average of 14.8 km (Labonte et al.1998). High fences, almost without exception, curtail normal dispersal patterns. A high fence in New York with 5 openings that allowed deer to move in and out of the enclosure delayed emigration of yearling male white-tailed deer (Nielsen et al. 1997).

Seasonal migrations of ungulates may be quite extensive, depending upon the species. White-tailed deer in Michigan, for example, moved an average of 8 km between summer and winter ranges (Van Deelan et al. 1998). High fences exclude confined animals from migratory ranges located outside of the fence. High fences may also disrupt migration patterns of free-ranging animals if high fences surround critical migratory range or block important corridors used by ungulates to access migratory ranges. In Wyoming, for example, a landowner blocked the migration pathway of a pronghorn herd with a fence, resulting in excessive winter mortality (H. Harju, Wyoming Game and Fish Department, personal communication). The fence design was subsequently altered to allow pronghorn to cross.

Habitat Impacts

Confining ungulates for wildlife management purposes is an extremely intensive and costly management technique. On properties where reduced ungulate density is a management goal, such as on 2 Texas Parks and Wildlife Department management areas, vegetative diversity within high fenced areas may be greater than habitats outside the fence. However, on those properties with higher ungulate densities, vegetative conditions within the enclosure may be reduced.

Within high fences, the interacting effects of confinement, high ungulate density and supplemental feeding, result in ungulate populations that can quickly exceed carrying capacity. Vegetative diversity and ecological health decline in areas where ungulate populations are allowed to exceed carrying capacity (Miller et al. 1992, Stromayer and Warren 1997, Miller and Wentworth 2000). Impacts of exceeding carrying capacity are not limited to ungulates. Excess herbivory has been shown to affect other species of wildlife that feed, nest, roost, or hide in lower canopy levels or dense grasses (Casey and Hein 1983, deCalesta 1994, McShea 1997). Many confined ungulate populations existing today subsist on overbrowsed or overgrazed ranges and supplemental nutrition is provided.

Vegetative diversity may be significantly higher inside high fences designed to limit ingress of animals into areas practicing density control through hunting. Many high fenced ranches in Texas could serve as demonstration areas for sustainable management of healthy white-tailed deer populations and native vegetation. Managed stands of native "brush" in southern Texas can rarely be improved upon as deer habitat and high fences help managers control the level of herbivory on native forages. When combined with habitat management and regulation of grazing pressure by domestic livestock, high-fenced ranches enclose some of the most ecologically diverse areas in Texas (S. J. Williamson, Wildlife Management Institute, personal observation).

Impacts on Non-ungulate Species

Few studies have documented impacts to other species of wildlife from high fences. Most small to medium-sized mammals can move under, through, or over high fences, except when high fence designs are employed specifically to deny access to predators (i.e., 8-cm +/-mesh sizes with the bottom buried). Clevenger et al. (2001) found that black bears, grizzly bears, and cougars easily climbed and crossed a 2.4-m-high fence designed to keep ungulates off highways in British Columbia. Coyotes frequently crossed by crawling underneath the gaps in the fence created by uneven topography (Clevenger et al. 2001). High fences rarely confine or exclude javelinas in South Texas (S. J. Williamson, Wildlife Management Institute, personal observation).

Birds are not confined by a high fence designed to confine ungulates, but may be susceptible to collisions with the fence. Baines and Summers (1997) and Catt et al. (1994) documented mortality of woodland grouse in Scotland caused by collisions with wire mesh fences designed to exclude red deer from forestry plantations.

Diseases and Parasites Associated with Confinement

Disease often tops the list of issues related to wild ungulate farming and ranching from the perspective of the wildlife manager (Samuel and Demarais 1993). Disease issues have been reviewed by Miller and Thorne (1993), and most are still valid. Managers of domestic and wild species have concerns about infectious diseases. Depending on circumstances, infectious diseases may impact animal agriculture due to direct morbidity and mortality and associated lost productivity and economic costs. Disease may result in restrictions on trade involving movement of animals or animal products possibly carrying pathogens, and some diseases may have public health implications. Additionally, disease may influence local populations directly (for example, outbreaks of hemorrhagic disease), and concerns about translocating pathogens may impact movement of wild ungulates for restoration and translocation purposes (e.g., testing requirements imposed on animals crossing jurisdictional lines, animal sourcing). Finally, questions may arise about the safety of venison for human consumption.

In general, most free-ranging and captive ungulates are healthy. However, infectious diseases are a concern whenever animals are maintained at high densities, thus facilitating transmission of pathogens. High density populations may also be subjected to nutritional, environmental, or social stressors, which may reduce immunocompetence (Griffin 1989). When animals of different species and from various sources are mixed, exposure of naive individuals to pathogens may increase their disease risk. When these circumstances occur in confined wild ungulates, there may be risks to free-ranging ungulates. The risk of transmission of diseases from captive ungulates to free-ranging ungulates (and potentially from free-ranging ungulates to captive ungulates) depends upon the management circumstances (e.g., fencing, geography, etc.), the likelihood of direct or indirect contact between the captive and free-ranging species, and the routes of transmission of any given pathogen. Some pathogens may occur in geographically limited populations of free-ranging ungulates; however, good management practices dictate that introduction of new pathogens into wild populations should be avoided. Our focus is on diseases of cervids and highlights those diseases that have the greatest potential to impact free-ranging ungulates in North America.

Chronic Wasting Disease

Chronic wasting disease (CWD) of cervids is a transmissible spongiform encephalopathy (TSE), which has similarities to several diseases of humans (kuru, Creutzfeldt-Jakob Disease [CJD], and variant CJD) and animals (bovine spongiform encephalopathy or BSE ["mad cow disease"] and scrapie of domestic sheep). These diseases apparently are caused by proteinaceous agents called prions that are devoid of nucleic acids (Prusiner 1982).

Chronic wasting disease recently was reviewed (Williams and Miller 2002). It was first recognized in the late 1960s as a clinical syndrome among captive mule deer at wildlife research facilities in northeastern Colorado (Williams and Young 1992). In 1977 CWD was determined to be a spongiform encephalopathy by microscopic examination of brains from affected animals (Williams and Young 1980). Shortly afterward, CWD was recognized among captive mule deer at a wildlife research facility in southeastern Wyoming; animals had been exchanged between the Colorado and Wyoming facilities over the years. Diagnosis of CWD in Rocky Mountain elk from these same facilities followed (Williams and Young 1982).

In 1981, CWD was recognized in a free-ranging elk in Colorado (Spraker et al. 1997). Subsequently, it was found in free-ranging elk in Wyoming and in free-ranging mule deer and white-tailed deer in both states. The known distribution of CWD in free-ranging cervids has expanded rapidly in recent years. It occurs endemically in southeast Wyoming and northeast Colorado (Miller et al. 2000) and portions of the panhandle of Nebraska (B. Morrison, Nebraska Game and Parks Commission, personal communication). It was recently diagnosed in deer in southwestern Wisconsin (T. Hauge, Wisconsin Department of Natural Resources, personal communication), southern New Mexico (K. Mower, New Mexico Game and Fish Department, personal communication), western Saskatchewan, western South Dakota (R. Fowler, South Dakota Department of Game, Fish and Parks, personal communication), and on the western slope of the Rocky Mountains in Colorado (M. W. Miller, Colorado Division of Wildlife, personal communication). The source of CWD in free-ranging deer in Nebraska may have been a game farm with CWD-positive animals (B. Morrison, Nebraska Game and Parks Commission, personal communication) and there may be a link between elk farms with CWD in Saskatchewan and CWD in free-ranging deer. Studies are ongoing to understand the epidemiology of these situations and to determine the degree of infection in the local free-ranging deer populations. Many states and provinces are currently conducting surveillance in populations of free-ranging deer and elk and over the last few years thousands of cervids have been tested and found negative for CWD.

Within the last 5 years, CWD has become a disease of considerable concern within the captive cervid industry in North America after its diagnosis in elk on game farms in Saskatchewan and South Dakota during 1996 and 1997, respectively. This was followed by recognition of CWD in elk on game farms in Nebraska, Oklahoma, Colorado, and Montana, and most recently in Kansas (L. Creekmore, United States Department of Agriculture, personal communication). The presence of CWD has lead to quarantine and/or slaughter of elk herds. The number of animals involved is large; over 8,000 privately owned elk in

Saskatchewan and 1,600 elk in Colorado have been or are scheduled for depopulation. Chronic wasting disease in the commercial elk industry forced the United State Department of Agriculture to declare CWD an animal health emergency (USDA 2001a) in order to obtain funding for indemnity to compensate owners of elk slaughtered to control CWD in the industry. A federal CWD management program for the captive cervid industries is currently being developed.

The origin of CWD within the captive cervid industry is not known and there are currently no known direct epidemiologic links to the free-ranging cervids in Wyoming, Colorado, or Nebraska. Chronic wasting disease in free-ranging cervids predates recognition of the disease in the captive elk industry. The epidemiology of CWD in the commercial cervid industry in North America is being investigated and the geographic extent will become better known as federal, state, and provincial control and monitoring programs are instituted.

Only 3 species of Cervidae are known to be naturally susceptible to CWD: mule deer, white-tailed deer, and Rocky Mountain elk. Subspecies of these cervids probably are also naturally susceptible, as was demonstrated by diagnosis of CWD in a black-tailed deer resident in a CWD endemic facility (Williams and Young 1980). Of concern to the captive cervid industry is the likelihood that other subspecies of *Cervus elaphus* (red deer, Manitoba elk, tule elk) also are susceptible to CWD.

Domestic livestock are not known to be naturally susceptible to CWD. A few cattle, sheep, and goats have resided in research facilities with CWD for prolonged periods without developing the disease. Three of 13 cattle developed CWD following intracerebral inoculation with an incubation period of between 24 and 27 months (Hamir et al. 2001). Cattle exposed to CWD agent via oral or contact routes remain healthy approximately 50 months post-inoculation, but these studies are planned to continue for a total of 10 years.

The specific routes of transmission of CWD are unknown. There is no evidence that CWD is a food borne disease associated with rendered ruminant meat and bonemeal, as was the case in BSE (Wilesmith et al. 1988). Occurrence of the disease among captive deer and elk, many of which were acquired from the wild, and field and model data provide strong evidence of lateral transmission (Williams and Young 1992; Miller et al. 1998, 2000). Maternal transmission may also occur; however, this has not been definitively determined. The epidemiology of CWD in free-ranging cervids is actively under study (Spraker et al. 1997, Conner et al. 2000, Miller et al. 2000, Gross and Miller 2001).

Lymphoid tissues associated with the digestive tract (tonsil, cervical lymph nodes, Peyer's patches, mesenteric and ileocecal lymph nodes) of affected deer and elk contain PrP^{res} (Sigurdson et al. 1999, Williams and Miller 2000), thus alimentary tract shedding may also occur in CWD. The TSE agents are extremely resistant in the environment (Brown and Gajdusek 1991); pasture contamination has been suspected of being the source of prions in some outbreaks of sheep scrapie (Greig 1940, Pálsson 1979). Observations strongly suggest fence-line contact and/or environmental contamination as the source(s) for the CWD agent (Williams et al. 2000). Concentration of deer and elk in captivity or in the wild by artificial feeding may increase the likelihood of transmission between individuals.

Modeling studies indicate lateral transmission among free-ranging cervids is necessary to maintain CWD at the prevalence observed in the endemic areas. The models also suggest that CWD has been present in free-ranging populations for >30 years (Miller et al. 2000). Maternal transmission may occur, but this route of transmission alone when used in the model was not adequate to maintain the disease at observed levels (Miller et al. 2000).

Currently there is no validated diagnostic for CWD that can be used on a large scale on live animals. However, because PrP^{res} can be detected in lymphoid tissues early in the incubation period before the animals are showing clinical signs (Sigurdson et al. 1999), biopsy of tonsil and use of immunohistochemistry has promise in a research setting (Williams et al. 2002, Wolfe et al. 2002). This testing requires that the animal be anesthetized to obtain the biopsy; thus, this technique is not suited to testing of large numbers of animals.

There is no known treatment for animals affected with CWD and it is considered 100% fatal once clinical signs develop. If an affected animal develops pneumonia, treatment with antibiotics might prolong the course of illness, but will not alter the fatal outcome.

Control of CWD is problematic. Designing methods for control or eradication of CWD is extremely difficult in the face of long incubation periods, subtle early clinical signs, absence of practical antemortem diagnostic tests, the extremely resistant infectious agent, possible environmental contamination, and our lack of understanding of the mode of transmission. Management currently involves quarantine or depopulation of captive CWD-affected herds. Two early attempts to eradicate CWD from captive cervid facilities failed; the cause of the failure was not determined, but residual environmental contamination following facility clean-up was possible (Williams and Young 1992). Management of premises after depopulation for CWD remains controversial. It is not known if these premises could pose a risk to free-ranging cervids. The United States

Department of Agriculture has developed a proposed program for management and eradication of CWD from the captive cervid industry (United States Department of Agriculture 2001b).

Management of CWD in free-ranging animals is even more problematic (Gross and Miller 2001). Long-term active surveillance to determine distribution and prevalence of CWD has been instituted to assist in evaluating changes over time and the effect of management intervention. Translocation and artificially feeding cervids in the endemic areas has been banned in an attempt to limit range expansion and to decrease transmission of CWD. Localized population reduction in areas of high CWD prevalence is being conducted on an experimental basis in Colorado (M. W. Miller, Colorado Division of Wildlife, personal communication) and on a 374-square-mile CWD eradication zone in Wisconsin (Wisconsin Department of Natural Resources 2002). Simulation modeling suggested that selective culling for CWD control must be initiated when prevalence is very low (<0.01%) to be effective in eliminating CWD (Gross and Miller 2001).

No cases of human disease have been associated with CWD and a recent World Health Organization consultation stated that CWD is not currently known to affect humans (World Health Organization 2000). Investigation of several cases of Creutzfeldt-Jakob disease in young people in the United States who had hunted or consumed venison did not reveal a link to CWD (Belay et al. 2001). However, in the absence of complete information and in consideration of the similarities of animal and human TSEs, the public health concerns remain one of the reasons why CWD is important for wildlife managers.

The question of commercial marketing of elk carcasses from CWD-exposed animals for venison is being examined. Following slaughter or depopulation of elk from herds with CWD, brains are tested by immunohistochemistry. Test negative animals have been passed for human consumption in some states, while in other states all carcasses from depopulated animals have been destroyed. The Centers for Disease Control and Prevention (E. Belay, Centers for Disease Control and Prevention, personal communication) have recommended that carcasses from CWD-exposed herds not go into the commercial venison market because of lack of "informed consent" on the part of consumers.

The presence of CWD in captive and free-ranging cervids is a serious wildlife management problem. Indemnity for depopulated cervids has just been made available in the United States and is being used to compensate owners of affected herds in Canada. Guidelines for management of captive herds with CWD are being developed by federal, state, and provincial animal health officials in consultation with the affected industries, public health officials, wildlife management agencies, nongovernmental organizations, and the public.

Implications for free-ranging populations of deer and elk are significant. Deer and elk should not be translocated from CWD-endemic areas. Surveillance programs are expensive for wildlife management agencies. The impacts of the disease on the population dynamics of deer and elk are not currently known. Modeling suggests that CWD could detrimentally affect populations of mule deer (Miller et al. 2000, Gross and Miller 2001), though effects on free-ranging elk are much less likely.

<u>Bovine Tuberculosis</u>

Bovine tuberculosis (TB) is a bacterial disease caused by *Mycobacterium bovis*. It has a relatively wide host range, including humans, domestic animals, and wildlife. Because of public health concerns, as well as the economic impact of the disease in domestic cattle, bovine tuberculosis has been the subject of a state–federal eradication program involving the United States Department of Agriculture, state departments of agriculture, and the cattle industry for many years. Bovine tuberculosis is nearly eradicated from domestic cattle and game-farm cervids in the United States.

Bovine tuberculosis in game-farmed cervids became a serious problem in North America during the 1980s (Stumpff 1982, Miller et al. 1991, Essey 1992*b*, Rhyan et al. 1992, Thoen et al. 1992, Haigh and Hudson 1993, Whiting and Tessaro 1994). After a gap of less than 10 years without a recognized outbreak of TB in elk, the disease was identified in game-farm elk imported to Canada that originated from Montana (Essey 1992*a*). This recognition resulted in Canada closing the border to importation of cervids from the United States and extensive testing of captive elk and deer in herds across Canada and the United States.

Control of the disease was difficult because of lack of government compensation programs at adequate market value for elk that were killed. Several states lost tuberculosis-free status when cattle became infected from contact with elk (Essey 1992*a*) or other cervids, resulting in considerable hardship to livestock producers in affected states. Game-farm cervids are now included in the Cooperative State–Federal Bovine Tuberculosis Eradication Program in the United States using the Uniform Methods and Rules for the Eradication of Tuberculosis in Cervidae.

Surveillance of free-ranging wild animals in areas adjacent to one affected game farm detected *M. bovis*-infected mule deer and coyotes (Rhyan et al. 1995, Whipple et al. 1997). Subsequent surveillance has not detected persistence of *M. bovis* in wildlife in this area. Bovine tuberculosis is not currently known to be present in populations of free-ranging elk in the United States (Williams et al. 1995), but it occurs in a herd of elk in Manitoba.

There are relatively few examples of maintenance of TB in populations of free-ranging wild ungulates (Clifton-Hadley and Wilesmith 1991, Clifton-Hadley et al. 2001). Sporadic cases of TB were reported in the earlier part of the century in white-tailed deer (Schmitt et al. 1997); these cases were thought to have been directly due to transmission from affected cattle to wild ruminants. At that time, TB was relatively common in cattle herds and populations of wild ruminants were generally low, thus decreasing the likelihood that the disease would be maintained as self supporting infections within populations of free-ranging ungulates. However, under conditions of increased wild ruminant density, TB can be a significant problem.

Bovine tuberculosis was reported in elk, bison, and moose from Elk Island National Park, Alberta, in the 1950s, but was not maintained among those species following population reduction (Corner and Connell 1958). It is, however, maintained in populations of free-ranging red deer and elk in New Zealand (O'Neil and Pharo 1995, Clifton-Hadley et al. 2001). The presence of TB in free-ranging white-tailed deer in Michigan is a serious problem.

Currently, TB is endemic in a dense white-tailed deer population in northern Michigan (Schmitt et al. 1997). The disease was perpetuated among these deer by the practice of winter feeding that greatly concentrated the deer, thereby increasing the rate of transmission. It has resulted in significant changes in how these animals are managed and has brought public health and agricultural agencies, as well as the Michigan DNR, into the business of disease management of free-ranging species. Considerable personnel and monetary resources are currently being expended in Michigan for surveillance and management, with eradication of the disease in free-ranging white-tailed deer the goal. The consequences of establishment of TB in additional free-ranging cervid populations would be serious (Thorne et al. 1992, Schmitt et al. 1997).

The clinical signs of TB in elk and red deer have been reviewed (Clifton-Hadley and Wilesmith 1991). Diagnosis of mycobacterial infection may be difficult (Clifton-Hadley and Wilesmith 1991, Rhyan et al. 1992, Rhyan and Saari 1995). Culture and identification of *M. bovis* is required for definitive diagnosis of TB.

None of the antemortem diagnostic tests are completely reliable in individual animals, but they are useful for detecting infected herds (Haigh and Hudson 1993). Diagnosis of *M. bovis* infection in game-farm cervids is by skin testing (single cervical test, comparative cervical test). These tests are conducted by an accredited veterinarian and require a 3-day holding period between injection and evaluation of the test. Some additional tests are approved for use in game-farm elk, depending on the state or province.

Bovine tuberculosis is transmitted primarily by the respiratory route. An infected animal coughs and expels bacteria and exudates in an aerosol. If a susceptible animal inhales the bacteria, colonies may form in the lung. High densities of animals increase transmission between infected and susceptible animals. Exposure may also occur orally from consumption of forage and feed contaminated with the bacteria, in which case, the bacteria probably first infects the tonsils or lymph nodes associated with the digestive tract. Concentration of animals around feeding troughs probably facilitates both aerosol and oral transmission (Clifton-Hadley and Wilesmith 1991). Calves may become infected by nursing dams shedding the bacteria in milk associated with lesions in the mammary glands.

The organism has a thick, protective, waxy outer coating and hence is relatively resistant in the environment. Organisms survive protected in feces for months, but under conditions of exposure to sunlight (ultraviolet light), such as on open pastures, fluctuations in temperature, and desiccation, the organism may only remain viable for days or weeks (Mitscherlich and Marth 1984, Jackson et al. 1995).

Though predators and scavengers serve as significant reservoirs of bovine tuberculosis in the United Kingdom, where European badgers are important (Clifton-Hadley et al. 1993, 2001), and New Zealand, where brush tailed possums and feral ferrets are free-ranging reservoir species (Morris and Pfeiffer 1995, Clifton-Hadley et al. 2001), no such wild reservoir is considered significant in North America. Wolves, coyotes, raccoons, black bear, and bobcat (Tessaro 1987, Whipple et al. 1997, Bruning-Fann et al. 1998, 2001) may become infected, presumably via consumption of carcasses of tuberculous ungulates.

Humans are susceptible to TB, though it is not nearly as common in humans as tuberculosis caused by *Mycobacterium tuberculosis*. Some humans in contact with game-farmed elk became infected and skin-tested positive (Fanning 1992, Stumpff 1992).

Meningeal Worm

Meningeal worm (*Parelaphostrongylus tenuis*) belongs to a small group of lungworms that are associated with connective tissues of the central nervous system and musculature of Cervidae. The biology of this parasite recently has been reviewed by (Lankaster 2001). Its usual definitive host is white-tailed deer. Meningeal worm is found throughout the deciduous forests of eastern North America and has not been recorded west of approximately 105° W. longitude. Intermediate hosts are terrestrial snails and slugs.

Meningeal worm is relatively innocuous in white-tailed deer, but it can cause a serious neurologic disease in many domestic and wild ungulates. Neurologic disease has been observed in naturally or experimentally infected caribou (Anderson and Strelive 1968), elk (Carpenter et al. 1973, Samuel et al. 1992), fallow deer (Pybus et al. 1992), moose (Anderson 1964), mule deer (Tyler et al. 1980), bighorn sheep (Pybus et al. 1996), pronghorn antelope (Anderson and Prestwood 1981), llama (Rickard et al. 1994), domestic sheep (Pybus et al. 1996), and domestic goats (Anderson and Strelive 1972).

Diagnosis of meningeal worm infection is by examination of feces for first-stage larvae using some version of the Baermann technique. The larvae of *P. tenuis* have a dorsal spine (dorsal-spined larvae). Recently developed techniques may assist in diagnosing *P. tenuis* infection by blood tests (enzyme-linked immunosorbent assays, Bienek et al. 1998, Ogunremi et al. 1999), but these have not been fully validated or used in the field.

Because larval shedding may be intermittent, experiments suggest that infected elk could go undetected by currently used diagnostic techniques (Welch et al. 1991). Samuel et al. (1992) successfully infected white-tailed deer with elk-origin larvae, proving that transmission from elk to deer is possible.

Treatment of white-tailed deer with ivermectin (an anthelminthic) is not effective in removing adult *P. tenuis.* The drugs may induce temporary cessation of larval shedding in deer feces (Kocan 1985, Samuel and Gray 1988), resulting in false negative fecal examinations. Thus, requirements for fecal examinations to detect *P. tenuis* infection for the purposes of excluding infected animals need to take the possibility of ivermectin treatment causing false negative results into consideration.

For meningeal worm to become established in a new area, first-stage larvae in feces of a definitive cervid (white-tailed deer or elk) must reach local terrestrial gastropods and develop to the infective stage. The snails and slugs must then be ingested by suitable ungulate hosts. Only a few gastropods are important intermediate hosts for meningeal worms (Lankester and Anderson 1968, Lankester and Samuel 1998); several of these are widely distributed across western North America, where *P. tenuis* does not occur.

The risk of accidental introduction of *P. tenuis* to susceptible ungulate populations in western North America through movement of deer and elk has generated considerable concern and controversy (Samuel 1987, Samuel et al. 1992, Miller and Thorne 1993). Meningeal worm could become established in western North America if the parasite were introduced in areas where there are populations of white-tailed deer, appropriate gastropod intermediate hosts, and conditions suitable for survival of the worms.

<u>Paratuberculosis</u>

Paratuberculosis, also called Johne's disease, is a bacterial disease primarily affecting the digestive tract of cattle, caused by *Mycobacterium avium paratuberculosis*. All bovids and cervids are considered susceptible to infection and disease caused by this bacterium (Williams 2001). This organism, which is distantly related to *M. bovis* (the cause of TB), also is quite resistant in the environment. It may persist in soil for a year or longer (Mitscherlich and Marth 1984), but it is relatively sensitive to exposure to ultraviolet radiation from sunshine, drying and high temperatures. Under natural conditions, it probably remains viable less than a year in the environment.

Chiodini et al. (1984) reviewed general features of paratuberculosis in ruminants and a more recent review covers the disease in small ruminants and deer (Stehman 1996). It is a disease with primary effects on the intestinal tract. High densities of susceptible animals contribute to transmission of this infection. The incubation period is prolonged and may take years. Thus, young animals are seldom clinically affected, though they may be infected, and the disease is primarily observed in mature animals.

Paratuberculosis has been reported in many species of wild ruminants (Williams and Spraker 1979, Chiodini et al. 1984), but in North America it is only known to be endemic in a herd of tule elk in California (Jessup et al. 1981), in a small population of Key deer in Florida (C. Quist and V. Nettles, Wildlife Health Associates, Incorporated, personal communication), and in several herds of bighorn sheep and mountain goats in one area of Colorado (Williams et al. 1979). Paratuberculosis has been maintained in tule elk at Point Reyes National Seashore, California (Jessup et al. 1981), for at least 20 years (Cook et al. 1997). These elk probably contracted the disease via contact with pastures contaminated by dairy cattle infected with *M. avium paratuberculosis* (Jessup et al. 1981). Rocky Mountain elk are susceptible to experimental infection by the oral route (Williams et al. 1983*a*), but clinical disease has not been observed in free-ranging elk other than at Point Reyes. This disease is of concern in the game farming industry, where it exists (Gilmour 1984, Griffin 1988, Haigh and Hudson 1993, Power et al. 1993), but the prevalence in North America is not known.

Diagnosis of paratuberculosis in the live animal is difficult, as is true of many mycobacterial diseases (Thoen and Haagsma 1996). There are several types of blood tests (ELISA tests, complement fixation tests, immunodiffusion tests) that measure antibody production and other blood tests (lymphocyte blastogenesis tests) that detect cell-mediated immunity. However, none of these tests are ideal and false negative and false positive results are possible. Culture of feces for the bacteria is a definitive method of diagnosis. This method is useful on a herd basis, but it lacks sensitivity in individual animals because of sporadic shedding of the organism in the feces of subclinically infected individuals. There are newer tests for detection of *M. avium paratuberculosis*, including radiometric detection (Collins et al. 1990, Cook et al. 1997) and molecular techniques (de Lisle and Collins 1995, Thoen and Haagsma 1996), but none are completely satisfactory.

Environmental conditions and animal behavior play a role in maintenance of this organism in free-ranging populations. The bacterium survives best under humid conditions with reduced exposure to sunlight (ultraviolet radiation). Thus, paratuberculosis is seldom a problem in dry, high-elevation environments. Because the organism is shed in feces, and transmission is via ingestion, behavior which concentrates animals, especially at a young age, will potentiate transmission of the organism.

The presence of paratuberculosis in herds of free-ranging wild ruminants is a management problem for several reasons. This disease may be fatal in a small percentage of animals, with a great many other animals having subclinical infections. These subclinical animals may shed the organism into the environment, thus serving as a reservoir of the organism for other susceptible animals. The biological effect of subclinical infection on individual animal performance is not known for wild species, but paratuberculosis is considered economically significant in domestic livestock. Herds of wild ungulates with paratuberculosis are usually not considered suitable as source herds for relocations, though quarantine protocols have been developed in an attempt to manage growing tule elk populations with paratuberculosis by transplantation. The presence of paratuberculosis in free-ranging ungulates causes conflicts with agricultural interests. The prevalence of paratuberculosis in captive cervids is not known.

A controversy exists as to the zoonotic potential of paratuberculosis. Some researchers have reported evidence linking *M. paratuberculosis* with Crohn's disease of humans (Chiodini and Rossiter 1996, El-Zaatari et al. 2001), but other researchers do not believe there is an association between these conditions (Van Kruiningen 1999). Even if such a relationship was confirmed, the possibility that humans would contract this disease from contact with wild ungulates or their feces seems unlikely.

Diseases of Lesser Concern

Diseases of less concern are thus categorized due to low likelihood for transmission from confined to free-ranging ungulates. The consequences of transmission, should it occur, probably would be less serious than for the previously discussed diseases.

Cervid adenoviruses. The importance and distribution of adenoviruses in cervids is currently unknown. A large outbreak of adenoviral hemorrhagic disease in California among free-ranging black-tailed and mule deer in the 1990s resulted in death of thousands of animals (Woods et al. 1996). Since then, sporadic cases of adenoviral infection have been diagnosed in captive moose and white-tailed deer elsewhere in North America. The epidemiology of cervid adenoviruses is poorly known but the virus (or viruses) is probably much more widely distributed than is currently recognized both among free-ranging and captive cervids. Based on an understanding of the epidemiology of adenoviruses in domestic livestock, it is likely that many animals may be exposed to the virus and only under stressful conditions is overt disease manifested.

Transmission is probably via direct contact and aerosol. There are no commercial diagnostic tests currently available for cervid adenoviruses. Humans and domestic livestock are not known to be susceptible to cervid adenoviruses. Methods to reduce direct contact between free-ranging cervids and alternative livestock would decrease the potential for transmission of cervid adenoviruses from captive to free-ranging cervids or vice-versa.

Cerebrospinal elaphostrongylosis. The biology of these nematode parasites has been reviewed by Lankaster (2001). These are close relatives of meningeal worm (*P. tenuis*). These worms are considered exotic to North America or are only found in geographically limited areas on this continent. Though there is considerable confusion over the taxonomy of these parasites, *E. cervi* is considered to be a parasite of red deer and moose and *E. rangiferi* is naturally found in caribou and reindeer. *Elaphostrongylus rangiferi* was introduced into Newfoundland, Canada, with reindeer originating in Scandinavia and is now established in free-ranging caribou. *Elaphostrongylus cervi* is a common parasite of red deer and was introduced from Europe into New Zealand when those animals were imported and released. Imported infected red deer have been detected in quarantine facilities in Canada.

In general, these parasites are innocuous worms that live in the skeletal muscles, but occasionally they cause disease in the lung, brain, and spinal cord when they migrate, which may lead to death of the normal host, as well as aberrant hosts such as domestic livestock and other species of cervids. The life cycle of the parasite involves slugs and snails as intermediate hosts. Diagnosis of infection is by examination of the feces for larvae but determination of the species is difficult unless the adult worm is recovered. This is similar to the difficulties encountered when diagnosing *P. tenuis.* Larvae may not always be found in the feces of infected animals because shedding may be low and intermittent, thus repeated testing is required. In addition, treatment with some anthelmintics will reduce larval production for a short time, but will not kill the adult worms, thus false negative diagnostic results may occur when testing fecal samples.

Giant liver fluke. The giant liver fluke *Fascioloides magna* is a natural parasite of white-tailed deer and elk, but it may infect many wild and domestic hosts. Pybus (2001) recently reviewed the biology of this parasite. It provides one of the earliest known examples of introduction of an exotic parasite with translocation of the host, when elk were introduced from North America to Italy in 1865. Since that local introduction, the giant liver fluke has spread into many areas of Europe, causing disease in native wildlife and domestic livestock. When found in high numbers, especially in an abnormal host, the worm can produce extensive lesions in the liver, which may result in death. Intermediate hosts are various aquatic snails, thus the distribution of this parasite is dependent upon adequate habitat (wetlands) to support the snail hosts in adequate numbers. In normal hosts, the parasites form cysts in the liver and eggs are expelled through the bile ducts and out into the environment with the feces. However, in abnormal hosts, including cervids other than elk and white-tailed deer, and in domestic livestock, particularly domestic sheep, the parasites continue to migrate in the liver, which may result in significant impairment of liver function, economic loss due to condemnation of livers at slaughter, and even death.

Malignant catarrhal fever. Malignant catarrhal fever (MCF) is caused by 2 bovid herpesviruses: ovine herpesvirus 2, the cause of “North American” or “sheep-associated” MCF, and alcelephine herpesvirus 1, the cause of “African” or “wildebeest-associated” MCF. African form MCF is considered an exotic disease in North America and African antelope that could be hosts to MCF viruses are regulated in most jurisdictions and are not found on game farms. Clinical MCF occurs in domestic cattle, bison, and cervids, though the species vary in their degree of susceptibility. The epidemiology of ovine herpesvirus 2 infection is still being studied, but it appears that transmission of the virus primarily occurs in association with lambing and contact with neonatal lambs. The potential for other members of the subfamily Caprinae to transmit ovine herpesvirus 2 is not clear. The role that wild ungulates play is far from understood (Li et al. 1996), and white-tailed deer may actually harbor their own MCF herpesvirus (Li et al. 2000). Cervids are usually considered “dead-end hosts” and there is no direct evidence that these species are capable of transmitting the virus.

Septicemic pasteurellosis. Septicemic pasteurellosis in wild ruminants is caused by several serotypes (A:2, A:3,4, B:1, B:3,4) of the bacterium *Pasteurella multocida* (Miller 2001). Disease results when bacterial infection involves the blood; damage to multiple organs follows and results in rapid death. Outbreaks of septicemic pasteurellosis have caused death of elk on the National Elk Refuge, Wyoming (Franson and Smith 1988), and other feedgrounds in Wyoming, and sporadic cases have occurred elsewhere. Septicemic pasteurellosis also occurs in domestic and wild bovids, but the species and serotype of the bacteria varies in different species. Septicemic pasteurellosis is sometimes incorrectly confused with “hemorrhagic septicemia,” which is an infection caused by certain serotypes of *P. multocida* and is considered an exotic disease in North America. Unapparent infection is probably common and the bacterium probably resides in the throat and tonsils. Outbreaks of septicemic pasteurellosis are associated with environmental stress, such as severe winter weather in situations of high density, which facilitates transmission among animals. Transmission is by direct contact and aerosol transmission.

Because the bacterial serotypes that cause septicemic pasteurellosis are probably widely distributed in both free-ranging cervids and confined ungulates and the occurrence of the disease appears to be directly related to environmental conditions and stresses, risk reduction includes managing animals to reduce stress.

Rangiferine brucellosis. Brucellosis caused by *Brucella abortus* (bovine brucellosis) is not known to occur among captive wild ungulates in North America. However, *Brucella suis* biotype 4 causes brucellosis in some populations of free-ranging caribou and reindeer in parts of Alaska and Canada. Thorne (2001) reviewed features of rangiferine brucellosis. Like *Brucella abortus,* the bacterium is transmitted from an affected animal to a susceptible animal via contact with aborted fetuses, placenta, fluids, and reproductive tract exudates. Cervids other than reindeer and caribou are susceptible to infection, and there is some experimental evidence that rangiferine brucellosis may be fatal in moose. There are no reports of rangiferine brucellosis being maintained in populations of cervids other than *Rangifer*. However, other cervids will develop antibodies that cross-react on serologic tests for bovine brucellosis, which may cause confusion.

Currently Unidentified Diseases and Exotic Diseases
It is important to realize that not all potentially serious pathogens and diseases of captive and free-ranging ungulates have been identified. New diseases (due either to new or newly recognized pathogens or to new species affected by pathogens because of changes in host range) are being found in diagnostic laboratories throughout North America with regularity and it is frequently not possible to predict if these new pathogens or new host–pathogen relationships will have significant impact on populations of wild animals or how we manage them. Obviously, managing for unknown pathogens is nearly impossible. Thus, it becomes very important to maintain disease surveillance in populations of captive, as well as free-ranging, species and to guard against artificially mixing populations of wild ungulates.

In addition, wild ungulates are susceptible to many highly infectious diseases of domestic animals that are classified as exotic by the USDA or CFIA in Canada. It is important to note that foreign animal diseases could affect wild ungulates. For example, wild ungulates in North America are susceptible to foot and mouth disease virus. Introduction of a foreign animal disease into free-ranging ungulates could have devastating effects on wildlife and agricultural industries.

Genetic Diversity and the Management of Wild Ungulates

The intensive management of wild ungulates commonly involves one or more of the following: the establishment of game-proof enclosures, translocations of different genetic stocks, and selective harvest. These actions may affect population demographics and census size and also the patterns of genetic variation as a consequence of altering the breeding structure, reducing the number of breeding individuals, constricting the reservoir of genetic variation in the population, and blocking the infusion of new genetic material. This review describes genetic variation and how it is measured, as well as important population genetic concepts relevant to the confinement and intensive management of wild ungulates. These topics include: effective population size, genetic drift and founder effects, genetic bottlenecks, inbreeding, gene flow and dispersal, effects of habitat fragmentation and dispersal barriers, hybridization and genetic introgression, outbreeding depression, effects of selective harvest, and the dilution of unique genetic stocks.

Genetic Variation in Natural Populations
Genetic variation in natural populations is present at many different levels, but is typically referred to at the individual or population basis. Heritable genetic mutations are the ultimate source of genetic variation, while recombination results in new arrangements of existing genetic material (Hartl and Clark 1997). Individual genetic variation is usually described by the percentage of loci at which the individual is heterozygous, while population genetic variation is characterized by percentage of polymorphic loci, number of alleles per locus, or expected heterozygosity assuming Hardy-Weinberg equilibrium (Nei 1973, Lacy 1997). Some genetic variation between individuals and populations is quantifiable by phenotypic differences, but much individual and population genetic variation must be visualized at the molecular level by comparing protein or DNA sequences (Falconer and Mackay 1996).

Individual traits, such as developmental stability, growth rate, metabolic efficiency, fertility, survival, and disease resistance, are probably influenced by heterozygosity (Allendorf and Leary 1986, Falconer and Mackay 1996). Thus, population genetic variation is important for long-term persistence of a population in the face of environmental change (Lande 1988). The likelihood of population extinction is influenced by genetic diversity, as evidenced by the high probability of extinction in genetically depleted populations, as well as the association between population genetic diversity and variables which induce extinction (Nunney and Campbell 1993). Our knowledge of genetic variation in natural populations has increased rapidly during the past 3 decades, primarily due to the development of easily identifiable genetic markers and automated analysis techniques. Modern population genetics focuses on understanding the origin, maintenance, and function of genetic variation in natural populations.

Natural populations differ in the level of genetic variability, as well as in the frequency and types of alleles present. Large mammalian taxa exhibit patterns of genetic variation on both broad and fine geographic scales, even when populations are apparently contiguous. Allozyme variation is associated with geographic location in white-tailed deer (*Odocoileus virginianus*, Smith et al. 1984, Sheffield et al. 1985, Carr et al. 1986, Gavin and May 1988, Karlin et al. 1989). There also is evidence that white-tailed deer populations are genetically subdivided on a microgeographic (<8-km^2) scale (Sheffield et al. 1985, Kennedy et al. 1987).

Genetic differentiation or subdivision over short geographic distances has been documented in many large mammals, including moose (*Alces alces*, Chesser et al. 1982), mule deer (*O. hemionus*, Cronin et al. 1991), and mouflon (*Ovis gmelini*, Petit et al. 1997). Geographic patterns of genetic variation also vary temporally (Scribner et al. 1997), probably due to demographic and environmental stochasticity.

Effective Population Size

Census size is an important factor determining population genetic variation, but the effective population size (N_e) actually governs the maintenance or loss of genetic variation (Wright 1931, Nei 1987). Effective population size is a complex concept that is usually described as the number of breeding individuals in a population. In reality, N_e for wildlife species is often much smaller than predicted due to fluctuating census and family size, sex ratio, mating system, migration, genetic drift, and other stochastic variation (Wright 1931, Nei 1987).

Genetic Drift, Founder Effect, and Bottlenecks

Another important factor influencing population genetic variation is genetic drift (Wright 1931). Genetic drift is a random process by which allele frequencies fluctuate between generations. Since the alleles present in the offspring generations are a sample of alleles present in the parental generations, the allele frequencies are affected by sampling variation between generations, with sampling variation increasing as the number of parents decreases (Falconer and Mackay 1996). These random fluctuations of allele frequencies are more severe in small populations, where the effect is intensified by unequal reproductive success among the few breeding individuals (Hedrick and Miller 1992). Since all individuals do not contribute equally to reproductive effort, some individuals substantially impact the genetic composition of subsequent generations, while others have little or no contribution. The long-term survival and fitness of small populations is threatened because genetic drift becomes more important than natural selection in their evolution (Lacy 1997).

Two situations where genetic drift may have a large influence on population genetic diversity are founder events and bottlenecks. The founder effect results from establishment of a new population by a small number of individuals (Nei 1987). A bottleneck occurs when a previously large population undergoes a severe reduction in size (Nei et al. 1975, Nei 1987). The genetic structure of the new population is dependent on the genetic variation in the founding individuals and their offspring. In each case, N_e is small and genetic variability is usually decreased in the new population. The effects of the genetic bottleneck may be long-lived, especially if population size remains small after the bottleneck, because new genetic variation will not accumulate for many generations. For example, species that have undergone known historical bottlenecks typically have little diversity within the major histocompatibility complex (MHC), a genetic system important in disease recognition and resistance. Contemporary populations of Przewalski's horse and Arabian oryx (*Oryx leucoryx*) were founded from <20 individuals and have little MHC diversity (Hedrick et al. 1999, 2000). A population bottleneck predating moose range expansion into North America contributed to low MHC diversity in present European and North American moose populations (Mikko and Andersson 1995). The MHC locus diversity in South African bontebok (*Damaliscus pygargus pygargus*), which underwent 2 severe population bottlenecks, was far less than in non-bottlenecked blesbok (*D. p. phillipsi*, Van Der Walt et al. 2001). Fitzsimmons and Buskirk (1997) found lower allozyme heterozygosity and fewer alleles per locus in 3 of 4 reintroduced bighorn sheep populations compared to the source population. The founding populations were small ($8 > n > 69$) and N_e remained low for 10–20 years post release.

Inbreeding

As population size decreases or populations become subdivided, inbreeding (mating between related individuals) is more likely to occur. Inbreeding increases the probability of 2 alleles at a locus being identical by descent from a common ancestor (Lacy 1997). An increase in the amount of inbreeding decreases heterozygosity, and individuals are more likely to become homozygous for deleterious recessive allele combinations, which are present at low frequencies (therefore rarely expressed) in large, random-mating populations (Falconer and Mackay 1996). Inbreeding depression effects that reduce survival have a greater effect on population extinction probability than on effects that reduce fecundity (Mills and Smouse 1994). Although inbreeding, even at low levels, probably has a greater effect on populations with low growth rates (Mills and Smouse 1994), inbreeding effects on fitness may be tolerable when inbreeding is gradual over time (Falconer and Mackay 1996). This is because there is opportunity to rid deleterious homozygous recessive alleles from the genome through selection (Falconer and Mackay 1996).

Most of our knowledge of inbreeding effects (e.g., reduced fitness and population viability) comes from laboratory animals and domestic livestock, but inbreeding probably affects wild populations similarly (Lacy 1997). Ralls et al. (1979) documented significantly greater juvenile mortality in inbred captive ungulates than in non-inbred captives. Fetal growth, maternal weight, and fetal number are positively associated with allozyme heterozygosity in white-tailed deer

(Cothran et al. 1983, Johns et al. 1996). Multilocus heterozygosity also is correlated to other traits presumably related to individual fitness in white-tailed deer (Smith et al 1982, Chesser and Smith 1987). Birth weight and neonatal survival are positively correlated with genetic variation in harbor seals (*Phoca vitulina*, Coltman et al. 1998). Coulson et al. (1998) observed heterosis (or "hybrid vigor") where mean allele length divergence at microsatellite loci was positively correlated with birth weight and neonatal survival in red deer, which they attributed to population mixing. Further research revealed sex-dependent differences in juvenile survival associated with inbreeding and outbreeding in red deer (Coulson et al. 1999). Inbreeding depression also affects lifetime breeding success in both male and female red deer (Slate et al. 2000). Inbred Soay sheep (*Ovis aries*) were more vulnerable to intestinal parasites and experienced reduced survival (Coltman et al. 1999).

Gene Flow and Dispersal

Some form of genetic exchange is necessary to link subpopulations and provide a continual source of new genetic material. Individuals which emigrate or disperse from one population into another will introduce their genetic material into the new population if they reproduce. This exchange of genetic material between populations is known as gene flow. Intrapopulation gene flow results in similarity of nuclear alleles and mtDNA haplotypes within a population. Subdivided populations experience reduced gene flow and an increased probability that the patterns of genetic variation will diverge (Honeycutt et al. 1999).

Gene flow is important in maintaining genetic variation in wild ungulate populations since the absence of gene flow would result in genetic substructuring due to social and behavioral factors. This is because individuals within populations may associate and disperse in nonrandom fashion or form social units based upon philopatry or coordinated dispersion of related individuals (Chesser 1991). A typical pattern in large mammals is female philopatry, which may subdivide populations along matrilines (Chesser 1991, Cronin et al. 1991, Mathews and Porter 1993, Mathews et al. 1997). For example, female white-tailed deer commonly are found in matrilinial groups composed of adult females, several generations of female offspring, and juvenile male offspring (Hawkins and Klimstra 1970, Hirth 1977, Mathews and Porter 1993, Mathews et al. 1997). Male dispersal acts to maintain gene flow between these population subdivisions. Yearling male white-tailed deer disperse from their natal groups and may establish new home ranges that are quite distant (Hawkins et al. 1971, Kammermeyer and Marchinton 1976, Nelson and Mech 1984, Dusek et al. 1989). Male white-tails also tend to expand their home ranges during the rut (Tierson et al. 1985), which increases the number of breeding opportunities with different matrilines and facilitates gene flow. An example of the importance of gene flow to diversity is Rocky Mountain bighorn sheep populations, where mtDNA diversity was maintained through metapopulation dynamics despite drastic reduction in census size and continuity of subpopulations (Luikart and Allendorf 1996).

Barriers to dispersal or factors affecting dispersal distance, such as high fences, can limit gene flow (Honeycutt 2000). Geography, climate, and habitat features may isolate populations and cause genetic divergence, contributing to population substructuring and partitioning of genetic variation. For example, island populations of white-tailed deer have significantly less genetic variation than mainland populations (Ellsworth et al. 1994*a*). Travis and Keim (1995) observed genetic differentiation between mule deer populations separated by the Grand Canyon in Arizona, and Cronin (1991) detected genetic differentiation between mule and black-tailed deer populations separated by the North American Cascade mountain range.

Artificial Dispersal Barriers

The genetic effects of artificial barriers to gene flow, such as exclosures, are difficult to predict because empirical documentation of the genetic effects of enclosures is lacking. However, habitat fragmentation and population isolation affect genetic variation in free-ranging populations, many of which are larger than enclosed populations. Habitat loss and fragmentation threaten genetic diversity of Asian wild cattle and buffalo species due to both isolation and reduction in census size (Heinen and Srikosamatara 1996). Gonzalez et al. (1998) documented genetic differentiation based on mtDNA sequences in populations of Pampas deer (*Ozotoceros bezoarticus*), which have been subdivided by habitat fragmentation since 1900. O'Ryan et al. (1998) demonstrated a significant relationship between genetic variation and population size in African buffalo (*Synercus caffer*) on fragmented game reserves using microsatellite markers. Luikart and Allendorf (1996) concluded that patterns of historic gene flow between Rocky Mountain bighorn sheep populations were affected by habitat fragmentation. Lee et al. (1994) observed greater mtDNA diversity in pronghorn antelope (*Antilocapra americana*) from Yellowstone National Park than in 28 other populations due to habitat preservation and robust historical census size in the Yellowstone population. Large genetic distances between contemporary gray wolf (*Canis lupus*) populations are the result of recent habitat fragmentation and reduced population size (Wayne et al. 1992). Old World wolves, with greater population dispersion, exhibit greater mtDNA subdivision than New World wolves (Wayne et al. 1992). In contrast, coyotes (*C. latrans*), which recently expanded their range, do not display genetic differentiation, probably due to extensive gene flow (Wayne et al. 1992).

Unlike habitat fragmentation, game fencing is an intentional barrier to gene flow. For example, net-wire fencing ≥2.5 m in height effectively restricts ungulate movements when properly maintained (McCullough 1979, Woolf and Harder 1979, Ozoga and Verme 1982). An enclosure of this type is essentially impermeable to immigration and emigration and may have the characteristics of a population bottleneck and/or founder event. Unless the enclosed area encompasses thousands of hectares, yearling males may not be able to disperse far enough from their natal group to avoid inbreeding with close relatives. Males breeding within the same matrilines for several generations would produce offspring with successively higher inbreeding coefficients. Enclosures also may alter the population breeding structure by concentrating individuals or influencing social structure. This occurs in white-tailed deer, whose normal breeding system involves pursuit and courtship of individual females by males (Hirth 1977). This breeding system probably results in a large male effective population size in natural populations which have balanced sex ratios and age structure. However, a single or small number of socially dominant males may monopolize breeding in enclosures, reducing the overall effective population size (DeYoung et al. 2002). Captive breeding facilities and other small, artificially enclosed populations are thus vulnerable to loss of genetic variation and viability (Honeycutt 2000).

Hybridization and Genetic Introgression

Hybridization may be described as mating between species, subspecies, or populations which differ genetically, while introgression occurs when there is genetic interchange between populations which hybridize via backcrossing of the hybrid offspring into either or both ancestral populations (Rhymer and Simberloff 1996). If population size is reduced, population genetic integrity becomes especially vulnerable to hybridization and introgression.

The genetic integrity of many Asian wild cattle and buffalo species is threatened due to hybridization with domestic ungulates (Heinen and Srikosamatara 1996). Expansion of exotic zebu cattle (*Bos indicus*) in western Africa, aided by advances in veterinary medicine and destruction of tsetse fly (*Glossina* spp.) habitat, threatens the genetic purity of trypanosomiasis-resistant taurine cattle (*B. taurus*, MacHugh et al. 1997). European bison (*B. bonasus*), North American bison (*B. bison*), and yak (*Bos grunniens*) show contemporary or historical genetic signatures of hybridization with domestic cattle (Polzhein et al. 1995, Schaller and Wulin 1996, Ward et al. 1999). Ward et al. (1999) found domestic cattle mtDNA haplotypes in 6 of 15 North American bison populations and 5.2% of all bison examined. Goodman et al. (1999) documented a hybrid zone in Scotland between native red deer (*Cervus elaphus*) and introduced Japanese sika deer (*C. nippon*). Where the 2 species are sympatric, up to 40% of individuals possess introgressed alleles. Though hybridization occurs at a low rate, substantial genetic introgression has taken place in the 30 years since sika and red deer became sympatric (Goodman et al. 1999). Lehman et al. (1991) documented coyote introgression into gray wolf populations in the north central U.S. and Canada due to recent expansion of coyote populations caused by changes in forest ecosystems associated with agriculture. Other studies also have documented coyote introgression into gray wolf (Wayne et al. 1992), red wolf (*C. rufus*), and eastern Canadian wolf (*C. l. lycaon*) populations (Wilson et al. 2000).

White-tailed deer (*O. v. texanus*) are expanding into areas of western Texas occupied by mule deer (*O. h. crookii*) due to invasion of woody species, which creates favorable habitat for white-tails (Wiggers and Beasom 1986). Early research indicated that the occurrence of mule deer–white-tailed deer hybrids in western Texas deer populations varied from 0 to 24% (Stubblefield et al. 1986). Carr and Hughes (1993) documented substantial hybridization between the 2 species in western Texas, with gene flow occurring predominantly from mule deer into white-tailed deer. Cronin (1991) observed introgressive hybridization of mtDNA from mule (*O. h. hemionus*) and black-tailed deer (*O. h. columbianus* and *O. h. sitkensis*) into white-tailed deer (*O. virginianus*) and widespread interbreeding of mule and black-tailed deer where the 2 species overlap.

Though the narrow zones of introgression may not threaten the genetic integrity of declining mule deer or expanding white-tailed deer populations in western Texas (Derr 1991), displacement of mule deer with white-tails and hybrids is symptomatic of ongoing habitat loss for mule deer (Wiggers and Beasom 1986). The reduced population size of mule deer may result in lost economic opportunities for private landowners in these regions (Carr et al. 1986, Stubblefield et al. 1986). In addition, there is genetic evidence for introgression of black-tailed deer (*O. h. columbianus*) genes into endangered Columbian white-tailed deer (*O. v. leucurus*) in the Pacific Northwestern United States (Gavin and May 1988). The patchy distribution of small Columbian white-tailed deer populations within a continuous distribution of black-tailed deer may presage dilution of the Columbian white-tail genome (Gavin and May 1988).

Outbreeding Depression

Outbreeding depression is a phenomenon similar to hybridization, which may occur when genetic stocks from different ecotypes interbreed (Templeton 1986). The hybrid progeny have reduced fitness due to differences in chromosome number, phenotype, or interacting gene complexes (Honeycutt 2000). An extreme example of outbreeding depression occurred during the restocking of ibex (*Capra ibex*) in Czechoslovakia. Ibex in the source populations, Turkey and the Sinai, bred at different times than the native ibex and resulting hybrids produced offspring during winter, driving the population to extinction (Templeton 1986). Captive white-tailed deer in Mississippi of northern lineage and northern-crosses experienced reduced survival versus southern deer, primarily due to hemorrhagic disease and pneumonia (Jacobson and Lukefahr 1998). Even where hybrids are not viable and genetic introgression does not occur, threatened populations receive no benefit from reproductive effort wasted on offspring that are not viable (Rhymer and Simberloff 1996), especially in species of k-selected large mammals where there is significant parental investment in few offspring.

Harvest

Harvest, which is essentially a form of artificial selection, is another factor influencing population genetic variation. Harvest plans may affect patterns of within- and between-population genetic diversity by altering population demographics, breeding structure, and N_e, especially under male-biased harvest (Ryman et al. 1981). Population genetic variation may quickly be reduced under some harvest regimes, and the effect is largely independent of population size (Ryman et al. 1981). Fitzsimmons et al. (1995) found positive correlations between horn volume and allozyme heterozygosity in bighorn sheep. The authors hypothesized that selective removal of large-horned males may decrease genetic variation and result in loss of fitness in small insular bighorn populations. Likewise, allozyme heterozygosity was positively correlated to antler size in white-tailed deer (Smith et al. 1982, Scribner et al. 1989, Scribner and Smith 1990). Harvest selection criteria based on antler size for young white-tails substantially decreased cohort antler size in some Mississippi deer populations (Strickland et al. 2001). Thelen (1991) suggested that certain elk harvest plans could affect population antler characteristics. Long-term differences in harvest plans between Himalayan tahr (*Hemitragus jemlahicus*) populations in New Zealand affected spatial distribution of sex and age classes (Forsyth 1999). Ellsworth et al. (1994*b*) recommended that harvesting of male white-tails be regulated to conserve interpopulation gene flow due to the male-biased dispersal pattern exhibited by white-tailed deer.

Unique Genetic Stocks

Besides monitoring individual and population genetic variation, molecular genetic techniques are useful for identification of unique genetic stocks or conservation units (Vogler and DeSalle 1994), which has management implications for large mammals. For example, white-tailed deer populations in the southeastern U.S. are a mixture of native stocks and those influenced by trapping and transplanting programs in the 1930s–1960s (Ellsworth et al. 1994*a*,*b*; Leberg et al. 1994; Leberg and Ellsworth 1999). If protection of native genetic stocks is desirable, further translocations into areas containing these stocks should be discouraged. Likewise, populations of pronghorn antelope in western Texas have been influenced by translocations, complicating the interpretation of conservation units (Lee et al. 1989, 1994). Bison herds with mtDNA from domestic cattle could threaten genetically pure bison herds (Polziehn et al. 1995, Ward et al. 1999). Besides affecting the genetic integrity of a population, hybrid individuals may complicate the legal status of threatened or endangered species (O'Brien and Mayr 1991, Rhymer and Simberloff 1996).

SOCIAL ISSUES

Ownership of Wildlife Resources

Public Trust Doctrine

"The guiding philosophy of our North American system of wildlife management is that endemic wildlife belong not to the individual, but to the people of the state and responsibility for managing that wildlife is entrusted to the governmental regulatory agency. This contrasts with the European system of wildlife management where wildlife is the property of the owner of the land on which the wildlife reside" (Stinson et al. 1999:8).

The concept that the government holds certain natural resources, including wildlife, in trust for the benefit of all people is known as the public trust doctrine. "The doctrine's roots extend back to Roman law, which held that by natural law, mankind held the common right to the use of resources such as air, wildlife, running water, and the oceans and their shores. In Common Law England the doctrine was transformed, so that ownership and disposition of rights to use these resources, particularly the beds of navigable waters, vested in the sovereign, and at least since the Magna Carta, these rights have been held in trust for the benefit of the people" (Meyers 1989:728).

Historically, the public trust doctrine has been found to apply most easily to water resources and their accompanying fauna, which in turn has led to the general recognition of the trust principle with respect to wildlife (Horner 2000). Meyers (1989) argued that, of all natural resources, wildlife is perhaps the most similar to water with regard to the difficulty of possession, and that there is a sound historical basis for extension of the doctrine to wildlife.

Meyers (1989) "was one of the first commentators to vigorously embrace the specific application of the public trust doctrine to wildlife management. He accurately noted that while there is little doubt that from the historical standpoint the public trust doctrine is applicable to wildlife, currently few, if any, states actively use the doctrine to protect wildlife or wildlife habitat. Most cases that have addressed the public trust in wildlife have focused on whether a state had the power to enact laws regulating the resource, and what might be the limits of such authority" (Horner 2000:27).

The application of the public trust doctrine to wildlife is deeply rooted in the history, beliefs, and court opinions of the United States, as has been well documented by Meyers (1989), Bean and Rowland (1997), Horner (2000) and others. In the 1842 case of *Martin v. Waddell*, "Chief Justice Roger Taney ruled that when the colonists became independent from England, property (including wildlife) formerly claimed by the king belonged to the state. This decision laid the groundwork for the doctrine of state ownership of wildlife" (Bean and Rowland 1997, as cited in Stinson et al. 1999:8).

In the 1896 landmark case, *Geer v. Connecticut*, the U.S. Supreme Court stated: "Whilst the fundamental principles upon which the common property in game rests have undergone no change, the development of free institutions has led to the recognition of the fact that the power or control lodged in the State, resulting from the common ownership, is to be exercised, like all other powers of government, as a trust for the benefit of all people, and not as a prerogative for the advantage of the government, as distinct from the people, or for the benefit of private individuals as distinguished from the public" (Horner 2000:40).

Horner (2000:40) concluded that the Supreme Court's decision in *Geer v. Connecticut* also "placed meaningful restraint on the ability of the government to privatize this resource" (wildlife), that "in the century that has passed since Geer, the courts have not backed off from the recognition of this trust relationship," and that "we have within the common law ample authority that the states, and the federal government where applicable, hold wildlife in trust for the benefit of all persons." Although the courts have consistently supported states' authority to enact laws regulating the wildlife resource, Horner points out they "have rarely addressed what obligations might co-exist with such authority."

In her comprehensive review of the application of the public trust doctrine to wildlife, Horner concluded:

1) "At the turn of the millennium, it can no longer be debated seriously that wildlife is held in trust for the public by the states. There is no need to 'extend' the doctrine to this 'resource.' The trust is there to be enforced" (2000:29).

2) "Not only is there ample rationale for the application of the doctrine to wildlife management, the states have had an unequivocal duty to manage wildlife in trust for all people since at least the Nineteenth Century" (2000:29–30).

3) "Because wild animals in their natural state are subject to neither private ownership nor actual state ownership, but 'belong' to everyone, claims of private property 'takings' as a result of wildlife regulation in the public trust fall flat" (2000:30).

4) "While the public trust doctrine has been universally accepted as a viable part of our legal heritage in the late Twentieth Century, it is anything but a working tool in the practices of public interest and conservation advocates across the nation" (2000:24). "The unfortunate fact is that most agency employees, or their governing boards or commissions, have never even heard of the public trust doctrine, much less understand it as any part of their mandate." However, "administrative officials cannot be expected to utilize and apply responsibly a legal principle that they do not know exists, and which appears nowhere in their agency mandate" (2000:42).

5) "The first step to making the implementation of trust principles a reality in the every day management of wildlife is the adoption of a recognizable statutory or constitutional directive" (2000:42).

<u>Achievements of North American Conservation</u>

The North American model of wildlife conservation has been described as the world's "most successful, economically productive, and most imitated system of wildlife conservation" (Geist 1988:17). Although he did not use the phrase, "public trust doctrine," Geist (1988:16) recognized the public trust concept when he stated that "the North American system of wildlife management is unique in that, with few exceptions, it makes the public both de jure and de facto owner of the wildlife resources." Geist (1988:16) also identified three primary policies basic to the success of wildlife conservation in North America: 1) the absence of market in the meat, parts, and products of game animals, shore- and songbirds; 2) the allocation of the material benefits of wildlife by law, not by the market place, birthright, land ownership, or social position; and 3) the prohibition on frivolous killing of wildlife.

Stinson (1999:9) summarized the remarkable North American conservation achievements reported by Geist (1988) as follows:

1) Restoration of decimated wildlife populations that remain wild and live in sustainable association with human culture.
2) Development of a large service and manufacturing industry centered around wildlife-related recreation.
3) A system of wildlife management based on state-employed wildlife managers responsible to elected representatives.
4) Development of conservation societies to fund and restore wildlife habitat and management activities.
5) Self-imposed taxation on behalf of wildlife (Pittman-Robertson, Dingell-Johnson, and Fish and Wildlife Conservation Acts of 1980).
6) Protection of extensive areas of wildlife habitat by state, Federal, and private conservation initiatives.
7) International treaties to conserve migratory birds and mammals.
8) Preservation of large predators as part of our North American wildlife heritage.
9) Development of a relatively inexpensive and efficient system of wildlife protection that allowed wildlife to recover and thrive.

Trends in Ownership of Wildlife Resources

"Private property rights or ownership of wildlife is an extremely contentious issue in the United States" (Teer 1998:67). Although the public trust doctrine dictates that wildlife is held in trust by the government for the benefit for the public, a basic tenet of United States property law is that landowners control access to their property and, thus, the public's access to wildlife resources on private lands.

Geist (1988) warned that attempts to switch wildlife from public to private control threaten to replace North America's highly successful system of wildlife conservation with one that, historically, has promoted neither the welfare of wildlife nor that of the public. Thomas (1997) reported that various interest groups have repeatedly attempted privatization of public lands in the United States. Although legal devolution of ownership of public lands from the State to the private sector has generally failed, transfer of ownership of wildlife on private lands has been more successful (Teer 1998).

"Wildlife conservation on private lands is evolving from regulatory to participatory management, from State to private control, from protectionism to sustainable use, and from free uses to all persons and societies to outright commercialization. These trends have had an impact on ownership of wildlife and its uses" (Teer 1998:67).

While some states, such as Wyoming, have aggressively protected the public's interests by banning privately owned game farms, other states have moved in the opposite direction. Teer (1998:67) reported that Texas "leads the nation in devolution of wildlife to the private sector." In contrast with former systems in which partnerships between users and owners were the norm, in Texas "wildlife is now being considered a commodity for sale and or exclusive use by the private property owner." "Wildlife is 'claimed' through such devices as high fences to contain large mammals," and landowners can "obtain permit(s) to capture deer from wild stocks, pen and breed them much the same as domestic livestock, and return them to the wild... " (Teer 1998:67). The reduction of a public resource to private ownership is a fundamental issue underlying the discussion of confinement of wild ungulates behind high fences for private or commercial purposes (Stinson et al. 1999). Geist (1992:558) argued that legalization of game ranching, of which the confinement of wild ungulates is a prerequisite, is in conflict with the underlying principles of the North American system of wildlife conservation. According to Geist, game ranching "transfers affected wildlife into the private domain," "is an abdication of public responsibility for wildlife," and "aims to create legal markets in venison and wildlife parts."

"Allowing private possession and sale of native wildlife in this manner requires a profound change in the guiding philosophy of North American wildlife management" (Stinson et al. 1999:9).

Hunting Ethics

Sportsmanship

One common theme concerned with hunting promotes a theory that modern man could not have evolved without the high protein meat diet provided by killing other animals. In a second common theme, it is argued that hunting is simply an immoral demonstration of mankind's baser instincts. The process of discussing and finding answers in this debate leads invariably to demonstration that ethical behavior is an essential component of hunting as we know it. Historically, the conversion from hunter–gatherer to an agricultural mode of existence has seen wildlife become the property of government, usually the king or the emperor. Hunting was reserved for royalty. "America is one of the very few countries on earth where the citizens, not landowners or the government, own the wildlife" (Smith 1993:108). This fact, unique in itself, has taken many species of American wildlife through a period of market hunting to near extinction, to the passage of game and fish protection laws, and to a wildlife abundance that is again unique in all the world. An interesting observation on this phenomenon not often recognized is "that laws alone were insufficient to stop the 'excesses of democracy' imposed on wildlife by mass participation in wildlife harvest. What arose to restrain these excesses was a philosophy called 'sportsmanship'" (Muth and Jamison 2000:843).

Defining sportsmanship and describing a satisfactory hunter ethic for modern conditions has not been an easy task or one that is likely to be concluded in our lifetime. Among the most widely published essays on this subject are those by the Spanish philosopher Jose Ortega y Gasset. He observes (1942:88) that hunting, "like every human activity, has an ethic which distinguishes virtues from vices." Aldo Leopold (1933:391) wrote, "Hunting for sport is an improvement over hunting for food in that there has been added to the test of skill an ethical code, which the hunter formulates for himself, and must live up to without the moral support of bystanders." More recently, Jim Posewitz (1994:16) has defined the ethical hunter as a "person who knows and respects the animals hunted, follows the law, and behaves in a way that will satisfy what society expects of him or her as a hunter."

That these discussions must continue and be expanded is perhaps demonstrated best by the success of a series of meetings that were started when Governor Stan Stephens and the state of Montana sponsored the first Governor's Symposium on North America's Hunting Heritage in 1992. The success of this initial symposium led to a series: a Second Annual Governor's Symposium on North America's Hunting Heritage in Pierre, South Dakota (1993); a Third Annual Governor's Symposium in Little Rock, Arkansas (1994); a fourth Symposium (1995) in Green Bay, Wisconsin; and a fifth Symposium (1998) in Hershey, Pennsylvania. In 2000, A Symposium on North America's Hunting Heritage was held in Ottawa, Ontario, and, in 2003, the sixth United States Governor's Symposium will be held in Austin, Texas.

Decker et al. (1993:23) clarified value and meaning: "The term 'heritage' tells us hunting is more than simply a particular form of outdoor recreation. You don't hear people, even the most avid participants, talking about our skiing heritage, hiking heritage, camping heritage, boating heritage, birdwatching heritage, or any other 'heritage' related to outdoor recreation." In truth and in fact, the reason hunting heritage is separated from all other outdoor endeavors is that hunting requires and imposes ethical standards on the participants. At every one of these meetings, professional wildlife biologists, outfitters, guides, wildlife managers, farmers, and ranchers explored the motivations and satisfactions of hunting and the methods needed to preserve the North American hunting heritage. Ethics, and the maintenance of ethics in hunting have been common themes through all the symposia.

Fritzell (1995:53) explained that ethics were important because, "to my mind, hunting will be tolerated by the American public only if it is perceived as having positive values that counterbalance the apparent negative ones." And what are acceptable values? Duda (1998:44) reported, "In general, hunting for food, hunting to manage game populations, and hunting for animal population control are very acceptable to Americans while hunting strictly for recreation or hunting for a trophy are much less acceptable." The degree to which some hunting is judged less acceptable is very often a consideration of fairness. This consideration has led to the development of the "fair chase" concept.

<u>Fair Chase</u>
Although native Americans had a hunting credo in which fairness was a major consideration (Nelson 1992), the origin of the term "fair chase" is generally credited to Theodore Roosevelt and the founders of the Boone and Crockett Club in 1887. The Boone and Crockett Club initially encouraged sportsmanlike methods of hunting, which by 1893 had developed into a "Credo of Fair Chase." Any trophy submitted to the Boone and Crockett Club's record book after 1963 had to be accompanied by an affidavit that the trophy was taken in Fair Chase (Ferguson 1964:22). Ferguson noted, however, "The Boone and Crockett Club realizes full well that sportsmanship cannot be legislated. The hunter who has a few days, intense desire for a trophy, and no scruples will not be detained by a rule in the book—nor even by a state law." However, as Nelson (1992:27) points out, "it would be a mistake to deny the existence of conservation ethics simply because we discover isolated cases where these ethics have been breached."

Posewitz (1994) provided a modern overview of hunter ethics with emphasis on the following points: 1) The ethical hunter knows and respects the animals hunted, follows the law, and behaves in a socially acceptable manner; 2) Fair chase is fundamental to ethical hunting because it addresses a balance that allows hunters to occasionally succeed, while animals generally avoid being taken; 3) Fair chase is important to hunting because the general public will not tolerate hunting under any other circumstance; and 4) Failure of high ethical standards and fair chase risks doing what is right for wildlife, risks the opportunity to hunt, and risks the self respect of the hunter.

<u>"Canned" Hunts</u>
Jose Ortega y Gasset (1942:49–50) explained, "It is not essential to the hunt that it be successful. On the contrary, if the hunter's efforts were always and inevitably successful it would not be the effort we call hunting, it would be something else." That "something else" is the "canned" hunt, in which the client pays the game rancher to kill a

specific type of animal under conditions where the probability of failure is reduced. As described by Lanka (1993:41), in some cases, "a 'hunter' picks and pays for a specific animal before the 'hunt' begins. In others, wildlife is baited to specific locations with feed or enclosed inside a small pasture before the 'hunt' begins. Many hunters and non-hunters alike find these types of practices unethical. Situations such as these could be used by anti-hunters in their attempts to ban all hunting." Causey (1992:54) is even more direct in questioning this practice: "Can shooting an actually or functionally captive animal enhance one's understanding of natural processes? Does... shoot[ing] exotic animals located for you by a guide honor your cultural heritage?"

Proponents of the game-farm industry and these practices tend to describe commercial game production "[as] divided into four categories: game farming, game ranching, game herding, and game cropping..." (Renecker 1993:20), and to imply a clear separation among categories. "Game ranching is the harvesting by hunting for a fee of wild animals.... Game farming, on the other hand, is the raising of domesticated deer or elk for the wholesale or retail meat market..." (Brown 1993:120). Geist (1988:18), however, explains that game ranching "differs from 'Game Farming,' a legal designation in Canada that denotes the raising of animals for viewing or live sale. Game ranching denotes the raising of big game to be killed for sale, or by paid hunting."

According to Renecker (1993:23), "Game ranching is an extensive type of enterprise that occurs on private or communal...properties... of at least 25 km^2 (6,178 acres) ... [from which] surplus animals [are] sold as breeding stock or slaughtered for meat. Owners could also exploit hunting opportunities on the ranch." Game farming, on the other hand, "occurs on private, deeded land that is again fenced to define ownership.... This strategy takes full advantage of all economic opportunities. For wapiti farming, this includes velvet antler sales, meat sales, and sale of breeding stock." Neither of these authors mentioned Seidel's (1993:109) estimate that the "acreage involved in an average 'farm' (50 acres) would not in most cases create a barrier to migration." Neither, obviously, would it provide much opportunity for a "wild" animal to escape harvest or a "hunter" to demonstrate any particular skill.

Leopold (1933:394) penned the relatively timeless observation that, "The recreational value of a head of game is inverse to the artificiality of its origin, and hence in a broad way to the intensiveness of the system of game management which produced it." Brown (1993), on the other hand, presented arguments in favor of hunting on elk ranches. Several of his points are presented here with contrasting views:

1) "Private ownership of elk or at least commercial gain from elk hunting can provide the impetus for habitat acquisition and improvement. Landowners are faced with a variety of options of using their land for farming, cattle ranching, mining, timber harvest or commercial development. Successful competition for hunting and non-consumptive enjoyment of elk will allow this alternate use to increase elk numbers and habitat" (Brown 1993:122).

In contrast, Geist (1985:597) described, "The notion of wildlife as a crop to be harvested by the public, with the emphasis on festive, wholesome enjoyment, is an American idea. It's a tradition rooted in history, an ideal to be cherished. There is danger in allowing wildlife to become a symbol of the rich, making hunting a frivolous pastime of the wealthy...." Lenzini (1992:47) concurs in writing, "Like politics, full-scale privatization of wildlife can strike at common use, smack of special privilege, and eventually put a public resource beyond the reach of the public."

2) "Elk ranching can help improve the public perception of hunting as a sport. Fee hunting is usually carefully monitored, and can propagate the notion among the non-hunting public that hunting is an ethical and safe sport, non-detrimental to the propagation of the species. Such evidence is necessary if the public is to continue to support the sport of hunting, and the costs associated with hunter education, law enforcement, game management and research" (Brown 1993:122).

In contrast, Posewitz (1994:58) stated "The concept of fair chase is important to hunting. The general public will not tolerate hunting under any other circumstance." Fritzell (1995:53) stated, "The motivation, attitudes and behavior of hunters will ultimately influence social acceptability of the practice." The slogan, "Real Hunters Don't Shoot Pets," used during debate over the Game Farm Reform Initiative in Montana in 2000, suggests segments of the public do not have a positive perception of fee hunting when it involves a "canned" hunt situation.

3) "Elk ranching can help maintain the hunting legacy in this country. Aldo Leopold, considered the father of wildlife management in this country, listed the tools of the wildlife manager as the ax, the cow, the plow, fire, and the gun. If we are to continue to use hunting as a tool of wildlife management, then we must propagate hunters. In our fast-paced society, few people have the time to scout out hunting territory and learn the biology and behavior of their prey. Public hunting areas are often overcrowded, and hunting experiences can be unpleasant and unproductive, especially for youth. Fee hunting can allow for a pleasant, ethical, safe and productive hunting experience, thus helping to ensure

that young hunters continue future participation in the sport" (Brown 1993:122).

Other wildlife professionals have a different perspective on fee hunting at an elk ranch. Peyton (2000:775) asked, "What lessons are learned by the young hunter placed in a blind to opportunistically harvest a game animal?" The Montana Chapter of The Wildlife Society (2000) believes that hunting on game farms reduces the concept of fair chase, is morally indefensible, and is degrading to both the shooter and the animal. Posewitz (1994:97) wrote, "The ethics of pursuing a trophy animal are closely tied to why we seek such an animal. If you hunt these animals because they represent the survivors of many hunts, and you respect that achievement, then you have selected a high personal standard. If, on the other hand, you pursue a trophy to establish that you, as an individual hunter, are superior to other hunters, then you have done it to enhance your personal status, and that crosses the ethical line." Geist (1989:176) observed that, "paid hunting must discriminate against the young or newly married or anyone with a modest income."

In his summary, Brown (1993:123) expressed concern over unlimited expansion of fee hunting; "If we become a society wherein only the wealthy can afford to hunt, then we incur the wrath of the disenfranchised hunters, and the general public will quickly lose interest in financial support of our hunting legacy. In that event, all of us...will be the losers, as will our precious elk herds."

Public Perception of Hunting

The right to hunt for meat has extensive public support, but opposition to hunting is considerable and a growing concern among hunters and wildlife managers (Kellert and Smith 2000:51). Organizations, such as the Humane Society of the United States and People for the Ethical Treatment of Animals, have media programs condemning sport hunting (Muth and Jamison 2000:845). While the field of environmental philosophy addresses the ethical and moral justifications for hunting, the anti-hunting movement continues to emphasize animal welfare and rights issues (Kellert and Smith 2000:51). In an increasingly urban society that lacks an appreciation of hunting as a recreational pastime or wildlife management tool, wildlife managers should be prepared to address the ethical concerns of anti-hunters and the general public (Kellert and Smith 2000:51). Sadly, "the American public has good reason to hold a dim view of the body collective known as hunters" (Kerasote 1993:50).

Posewitz (1993) called for an ethical agenda to improve hunting management, hunting practices, and hunting's image. Aasheim (1994) agreed that image is a common problem for the North American hunter. Possible courses of action intended to change that image may be difficult. Holsman (2000:808) suggested that "hunters often hold attitudes and engage in behaviors that are not supportive of broad-based, ecological objectives." Changing such attitudes and behaviors could be valuable because "an exhibition of stewardship among the hunting community may earn the respect of the non-hunting public" (Peyton 2000:777). Public perceptions within a democracy are critical because the majority perception could determine the future legality of hunting (Hayden 1992).

Commercialization and Domestication of Wild Ungulates

One of the recurring philosophical and legal questions concerning ungulates behind fences involves "wildness" versus domesticity. This is not simply a question of semantics because the definition often carries substantial legal and management implications. Free-ranging native wild ungulates are public property, and management and regulatory responsibility usually reside with a state wildlife agency. As domestic livestock, ungulates become a private responsibility, and the regulatory responsibility often resides with a department of agriculture.

According to the Fact Sheet of The North American Elk Breeders Association (Rich 1993:8), "Alternative livestock by common, academic and legal definition are agricultural resources that should be managed by the departments of agriculture or jointly with the departments of wildlife (natural resources)." In support of this definition, Rich (1993:8) cites 2 other publications: "a species is domestic if both reproduction and the habitat critical for reproduction are under human management. It is therefore semi-wild or semi-domesticated, if only one of the elements is met and wild if neither is met..." (Prescott-Allen and Prescott-Allen 1986), and "domestic animals are husbanded rather than hunted, produced rather than procured" (Hudson 1989).

Kahn (1993) contrasted the vast differences between "domestic" and "wild" ungulates, stating that the domestication process takes thousands of generations to facilitate the changes in behavior, conformation, color, and temperament necessary to distinguish domestic animals from wild animals. Most of the elk in captive situations in Colorado came from the Yellowstone area during the past 50–75 years. Croonquist (1993) and Dratch (1993) stated that captivity does not make elk into domesticated animals.

Lanka (1993:36) quotes Van Gelder (1979), who defines domestic animals as "populations that, through direct selection by man, have certain inherent morphological, physiological, or behavioral characteristics by which they differ from their ancestral stocks." Lanka (1993:38) also notes that, "Judge William A. Taylor of the Eighth Judicial District, State of Wyoming, ...ruled that confining wildlife in an agricultural setting does not by itself make them domestic."

A major impetus for expansion of the game farm industry in North America has been to diversify production on agricultural land when income is already restricted by acreage limitations on some crops and by relatively low prices for traditional domestic livestock. Despite the substantial initial investment for fencing, some landowners are attracted to the range of commercial products apparently produced by "alternative livestock" ungulates held behind fences.

There are 4 primary products in the game-farm industry: velvet, meat, breeding stock, and shooter bulls. Typically, the benefits of raising ungulates for venison are presented to the prospective investor as summarized by Brown (1993:121): 1) There is an economic market niche in this country for venison production, 2) Venison itself is a healthy product, 3) Very often, deer and elk are more efficient users of land than are more traditional livestock, and 4) Deer farming allows farmers and ranchers to diversify.

The general experience of many western game ranchers is that the venison market niche is very limited and mostly filled by imports from New Zealand. Brown (1993:122) continues with the observation that "my personal feeling about deer and elk farming is that the public will accept the production of venison from exotic animals much more readily than they will native species."

Lacking a strong market for venison, Rich (1993:2) admits that "Most elk farms today supply the velvet antler market, generally for export." As described by Renecker (1993), most of the world's velvet antler production comes from maral (*Cervus elaphus maral* and *C. elaphus sibiricus*), elk, red deer, sika deer (*C. nippon*), and reindeer (*Rangifer tarandus*).

Korea was once the major importer of North American velvet, but both Korea and Japan have prohibited imports following the CWD outbreaks in Saskatchewan and Colorado. North American producers are further isolated by marketing methods mostly controlled by Pacific Rim buyers. Some recent exception to this pattern has been the FDA approval of velvet pills produced in Oklahoma under the brand name *Nature's Force Velvet Capsules*, and in Minnesota as *Natural Velvet Capsules.* In both cases, as near as it is possible to determine in the advertising, the product has been approved for a single game farm rather than an industry cooperative or some more efficient operation.

Even as the venison and velvet markets have proved to be somewhat illusory, an already declining market for breeding stock has been impacted by a United States Department of Agriculture declaration of CWD as an animal health emergency (United States Department of Agriculture 2001a). Interstate, and even intrastate, movement of animals has been severely restricted or banned by some state regulations. Prior to this ruling, interstate shipment was significant in the development of the captive cervid industry in the northern United States. For example, a total of 936 deer and elk were shipped out of Michigan between 1997 and 1999 (Coon et al. 2002). Shipments of deer and elk into Michigan originated from Missouri, Wisconsin, Ohio, Minnesota, and Canada (Coon et al. 2002). If sale of live animals is limited, for many game farmers, the only remaining potential income source is selling the opportunity for shooting the enclosed animals.

Ecological Stewardship

Wildlife managers recently have begun to discuss the ethics of wildlife management as practiced in the modern world. A recent issue of *Wildlife Society Bulletin* carried a series of introspective papers examining and questioning relationships among hunters, trappers, and wildlife managers.

Leopold (1933) saw game management as an integrating profession in which all facets of ecological systems received consideration and yet, as the twentieth century came to a close, arguments were presented that the wildlife profession concentrated too much on consumptive use and control of populations (Wagner 1989). Organ and Fritzell (2000:785–786) agreed that, "hunting is typically marketed as an effective way to control wildlife populations and an important source of revenue for conservation programs. This marketing approach has a subliminal emphasis on killing and an overt emphasis on generating funds that are inconsistent with the historical development of sport hunting." Geist (1991) emphasized that wildlife should be killed only for cause, a concept that prohibits waste and encourages subsistence.

"In many states, game management programs are being challenged by concepts like 'ecosystem management,' 'biodiversity,' and 'conservation biology'" (Peyton 2000:774). The reason for this challenge is that, "In the enthusiasm to produce a wildlife surplus and then to harvest it, a critical ingredient often missing is the relationship of the hunter with the hunted and the ecosystem involved. That relationship...is essential to fostering an effective and well-informed sense of ecological stewardship among the hunting community. Farmers and hunters who are focused on maximizing production of cattle or deer and who do not understand or do not care how their product depends on and impacts the ecosystem will make poor stewards of the land

and its natural attributes" (Peyton 2000:777). "It is time to examine our own professional efforts in wildlife management to determine whether we are encouraging stewardship or simply promoting a form of agriculture among hunters..." (Peyton 2000:778).

The public perception of wildlife management, in some respects, parallels that of the professionals, but with far less introspection. Although wildlife biologists may understand Leopold's (1949) statement that conservation is a state of harmony between men and land, Peyton (2000) questioned how well the statement is understood and accepted by our consumptive wildlife user partners. Lenzini (1992:47) observed that a "startling number of citizens have lost all real connection to the land," and as a result, "it is regrettable that wildlife management is being politicized. Leopold set out in the 1920s to establish wildlife management as a professional discipline, some say a science, possessing a structure of its own. Today the principles of that discipline are succumbing to the belief that nothing matters beyond politically desirable results."

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THE WILDLIFE SOCIETY

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The Wildlife Society is the association of wildlife professionals dedicated to excellence in wildlife stewardship through science and education. The goals of The Wildlife Society are to: develop and maintain professional standards for wildlife research and management; enhance knowledge and technical capabilities of wildlife managers; advance professional stewardship of wildlife resources and their habitats; advocate use of sound biological information for wildlife policy and management decisions; and increase public awareness and appreciation of wildlife management. The Wildlife Society, founded in 1937, is a nonprofit organization whose members include research scientists, educators, resource managers, administrators, communications specialists, conservation law enforcement officers, and students from more than 70 countries.