Interagency Coordination among Wildlife Management Agencies
in the Presence of Source-Sink Population Dynamics

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ABSTRACT

Including Alaska, more than one-fourth of the US land area is federally owned. Studies of the political economy of this land management are underdeveloped, and our understanding of transboundary coordination between different units is virtually nonexistent. Transboundary issues are especially important for managing mobile resources (wildlife) or resources with externalities (timber and watersheds). This paper provides a first cut at understanding these issues with a model of agency decisions against a background of population biology. The paper defines agency objective functions in wildlife management in terms of mandates (or decision rules), including non-intervention, population recovery, and sustainable harvest. Combining mandates and populations yields a large number of possible transboundary cooperation problems, several of which I analyze in depth. The model yields insights into why transboundary cooperation within and between the US and Canada has been successful for migratory and anadromous species such as salmon, elk, and caribou, but unsuccessful in managing most endangered species and game animals. It also explains success or failure in state-federal and state-tribal coordination problems as well as large-scale management challenges such as the Greater Yellowstone Ecosystem.

Political boundaries do not usually coincide with ecological boundaries. Indeed, political boundaries are often chosen in a way that divides ecological units. Many boundaries divide watersheds by following rivers, or divide alpine ecosystems by following mountain crests. Other ecologically-irrelevant boundaries reflect arbitrary considerations such as straight lines or the vagaries of history. As a result, many land managers find themselves unable to reach their wildlife management goals unless they cooperate with the managers of adjacent land.

Professionally and intellectually coherent management of such an ecosystem requires coordination among the agencies involved. Despite the incentives for it, such trans-boundary cooperation (TBC) has proved to be elusive. As a result, professional biologists and managers regularly lament the lack of coordination, and call for greater interagency cooperation at the ecosystem level. Going beyond the purely hortative, they may also start new cooperation initiatives, only to be frustrated when they fail. As conservation biologists and others in the field increasingly realize, a failure to consider fully social, cultural and political factors provides an important part of the reason (Agee and Johnson 1988; Bawa et al. 2004).

Understanding both the successes and the failures really requires a social-scientific response to the problems that agencies face. Our goal should be a framework that can account for both successful and unsuccessful coordination.

This paper develops a model based on two variables: the characteristics of the resource and the goals of the manager. Sometimes adjacent managers have mandates that complement each other well, while others do not. At the same time, we must recognize that the same agencies might cooperate in one place but fail to cooperate in another, a mix of behavior that looking simply at conflicts in legal mandates (i.e., Keiter 1988) fails to explain fully.

To account for this variation in the success of TBC, the theory needs variation in the object of policy, that is, in the biological processes that agencies try to manage. I provide this with a simple BIDE (birth-immigration-death-emigration) model of source-sink population dynamics, which I contrast with migratory populations. Specifically, a given piece of land ("habitat") could be a "source" that produces more wildlife than it can sustain, with the surplus dispersing elsewhere; a "sink" that imports these surpluses; or a migratory habitat that is used only part of the year. Managers might seek to maximize the number of animals, the harvest of animals, let natural processes work, or pursue some other goal. Management goals that complement each other well with one mix of habitats would work less well with other wildlife populations.

In short, this paper provides a simple theory examining how management goals in population sources and population sinks interact to make interagency coordination either more or less likely. The analysis here is foundational, establishing an analytical framework for investigating the problem as opposed to rigorous and exhaustive proof of results. Those will come, I hope, in future papers–but the many permutations of agency mandates and population structures requires a gradual approach here.

Fragmented ecosystems and the problems of TBC

Political boundaries do not normally coincide with ecological boundaries because political decision-makers and ecological philosopher-kings maximize different goals. The political motivations for creating reserves explains why reserves of all types fail to protect entire ecological units such as the Greater Yellowstone Ecosystem (GYE) in the United States (Clark et al. 1991). This suboptimality–why societies don't draw better boundaries in the first place–provides the analytical background to the problem of TBC, which seeks to address this suboptimality.

Imagine a reserve whose boundaries are chosen for strictly ecological goals, subject to some cost for the land. At some point the marginal cost of adding more land to the reserve outweighs the benefits. Now imagine drawing the boundaries for the same reserve, but with political actors playing some role in the process. There is no reason to believe that the marginal political costs and benefits of each parcel would exactly coincide with the previous ecological decision. In fact, the two processes would generically *not* yield the same boundaries for any set of utility functions.[1] For example, the existence of any private benefits to a potential parcel would produce lobbying against including it in the preserve. The resulting divergence between social and private net benefits would tend to work against including the marginal parcel if the political decision-maker includes both sets of benefits in her utility function.

This claim that reserves will generally be too small for their goals should generalize across multiple types of reserves. A marginal parcel excluded from Yellowstone National Park's preservationist mandate may well be included in the Gallatin National Forest, where multiple use prevails. Farther out from the national park, however, forested bottomlands may well yield too many private benefits to be included in Gallatin NF–though, again, such lands provide valuable

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[1]I have in mind here an ecological decision rule that maximizes utility from a set of ecological goals $U_E = f(e_1,\ e_1 \ldots e_n)$. This will have a different optimum than a function that includes political objectives such as $U_P = f(e_1,\ e_2 \ldots e_n;\ p_1,\ p_2 \ldots p_n)$. Note that introducing even a single political objective should (generically) yield a different optimum. The closest analogue to this framework that I have found is Lueck (1991).

seasonal habitat for many species. The end result of this logic is the patchwork of the GYE and many other preserved landscapes. In the US, each agency then manages according to its own mandate up to its own boundary, without buffers or transitional management zones.

One implication of this political economy of reserve creation is that parks will tend to concentrate in “worthless lands” (Runte 1979/1987) or, more specifically, lands that are worthless except when producing park-related values such as tourism. A second implication is that reserves will generally be too small to meet their ecological goals. Great Smoky Mountains NP, for example, does not include the lower-elevation lands that whitetail deer, black bears, and other species use on a seasonal basis, because this real estate was historically valuable for farming and is now valuable for recreation and tourism. This results in regular human-wildlife conflicts in adjacent communities such as Gatlinburg, Tennessee (Brown 2000: Epilogue).

For several decades, island biogeography theory has provided an important tool for thinking about the biological consequences of these too-small reserves. Animals with relatively large habitat requirements, such as large and meso-carnivores, tend to be found in “islands” such as national parks, isolated from one another (Newmark 1987). Both gray wolves and grizzly bears, for example, are found in Yellowstone and Glacier NPs but are not established in the approximately 400 miles of territory between these parks. These parks therefore face a steady decline in genetic diversity, with eventual problems of inbreeding, as is found in the isolated wolf population on Isle Royale NP (Wayne et al. 1991). As a result of inbreeding, disease or other threats, each of these “island” populations faces a risk of local extinction because it cannot be replenished from a larger area in the case of, say, widespread wolf cub deaths from parvovirus in Yellowstone in 2005 or Isle Royale in the 1990s (Hamashige 2006; Peterson 1995). Even large-scale protected areas such as national parks may be insufficient. For example, Canada's southern national parks are too small, and experience too many visitors, to sustain minimum viable populations of large carnivores (Landry et al. 2001). The same result holds for smaller species as well as the large ungulates and carnivores. For example, European butterflies that depend on one or two types of habitat–such as forest, wetland, grassland or fen−have declined much more precipitously than species with a more diverse range, as many of their habitats have become too small to support a viable population (van Swaay et al. 2005).

Either creating a larger protected area, or providing corridors between populations that allow mixing of gene pools within a larger metapopulation, are therefore essential for the long-term health of these populations (see Bennett 1999). This is especially true for carnivores that tend to have much larger home ranges and wider dispersion rates, and prey will have to be available in corridors if predators are to find them attractive (Harrison 1992). The facts that ungulates exploit buffer zones between carnivore territories, and spread themselves out in a way to maximize carnivore search time (Mech and Peterson 2003), add to the complexity of the management problem across fragmented regional landscapes. In all these cases, TBC can play an important role in expanding protected habitat, developing wildlife corridors, coordinating predator-prey balance, and improving management overall.

The GYE, with about 25 relevant agencies, provides an example of the potential scope for these TBC problems when viewed in ecosystem terms. The story is similar when we look at individual species. for example, The Central Idaho Restoration Area for the gray wolf encompasses lands of the Bitterroot, Boise, Clearwater, Lolo, Nez Perce, Panhandle, Payette, Salmon, Challis, and Sawtooth National Forests as well as the Nez Perce Indian Reservation.

The restoration program was run by the US Fish and Wildlife Service (USFWS), who hired the Nez Perce IR as its lead management agency because of heavy political opposition in the state governments of both Idaho and Montana (Mack et al. 2002). Wolf management in the northern Rockies now involves five federal agencies, three state wildlife departments, seven Native American tribes, and land management agencies at the federal, state, tribal, and local levels of government (Fritts et al 2003).

Needless to say, these management units do not have common goals. Some manage for multiple use, including consumptive uses such as hunting, while others focus on non-consumptive uses such as tourism or existence values. These management units also operate in very different political settings, whether a purely local constituency (Nez Perce or Blackfoot IR), an entire state's hunters (Idaho's Fish and Game Department Wyoming's Game and Fish Department), a more mixed state constituency (Montana's Department of Fish, Wildlife & Parks), or a national constituency interested in tourism development and endangered species conservation (National Park Service [NPS], USFWS). The success of TBC in such settings varies considerably, as the next two sections analyze.

Population dynamics

Though the primary focus of this paper is on how management mandates affect TBC, this problem requires an understanding of the underlying policy problem, that is, the behavior of wildlife populations across multiple jurisdictions. TBC can be challenging even across similar habitats, but the issue presents more problems when reserves manage different kinds of habitats or populations.

For this reason, I will use a basic model of population dynamics applied to the topic of "source" and "sink" habitats (Pulliam 1988). The basic idea in this model is that some wildlife populations live in habitats that are sufficiently productive to yield a surplus of animals each year. Finding insufficient resources for their own consumption and reproduction, this surplus emigrates to other habitats nearby. Some of these lands are not productive enough to support a population by themselves, but this influx of animals suffices to maintain their population. For this reason, the ranges of in-migration are known as population sinks, while the surplus-generating ranges are known as population sources.[2]

The main variables in this model stem from this basic setup. A population has some rate of birth (recruitment) and death. In the two-habitat setting, we also wish to keep track of the immigration into and emigration out of each habitat. These variables–birth, immigration, death, emigration–give these the name of BIDE models. The key concepts are shown in Table 1.

The population in any habitat i will be stable when $b_i + i_i - d_i - e_i = (BIDE)_i = 0$. This is *not* an equilibrium condition, but a possible outcome. For most purposes, it is also a desirable outcome, as in the case of seeking sustainable harvest levels. For this reason, one of the goals of the political analysis below is to determine (1) which., if any, habitats maintain a stable population size under various management rules; and (2) how the presence or absence of TBC affects population (in)stability.

[2]More complex, but not incompatible models now hold sway in population biology, emphasizing multiple habitats of variable productivity in a fragmented landscape (i.e., Rodenhouse et al. 1997); see Bennett (1999) for an accessible introduction.

Table 1
Basic BIDE Model

Concept	Each habitat	Aggregate	Notes
births	b_2	$B=\sum^{m}_{j=1} b_1$	
deaths	d_2	$D=\sum^{m}_{j=1} d_2$	
immigrants from k to j	i_{jk}	$i_2=\sum^{m}_{k=1} i_{jk}$	
emigrants from j to k	e_{jk}	$e_2=\sum^{m}_{k=1} e_{jk}$	$\sum^{m}_{i=1} i_2 = \sum^{m}_{i=1} e_2$

Given this basic setup, it is easy to define the concepts of "source" and "sink." A source experiences more births than deaths and emigration exceeding immigration. A sink is characterized by more deaths than births, and immigration greater than immigration. These conditions are listed in Table 2.

Table 2
Definitions

Source	$b_1 > d_1$ AND $e_1 > i_1$
Sink	$b_2 < d_2$ AND $e_2 < i_2$

The basic events of birth, death, emigration and immigration take place in a particular order, often associated with particular seasons. The sequence for most of the large animals that pose the most politically salient management problems includes fall mating, winter die-off, spring births, and summer dispersals. This is overly complex for our analytical purposes, since some portion of pregnant females will die in the winter and therefore fail to reproduce in the spring. It is therefore easier to put the seasons out of sequence a little, as shown in Table 3. Here, we count the number *n* of animals in the fall and treat spring reproduction as the next stage.

Reproduction simply adds a fraction to the population, depending on a rate of reproduction (β). After this comes a die-off, which may affect adults and juveniles differentially. I model the survival rates of each (P_A and P_2), which means that death rates are defined in terms of $(1-P_A)$ and $(1-P_2)$. Note too that this distinction between adult and juvenile death rates effectively, if indirectly, captures the problem of winter deaths of pregnant females. Finally, any surplus animals disperse.

Table 3 shows this part of the model. It also assumes that there is a maximum number of breeding sites, such that any population greater than *n* will reproduce at the same rate as a population of n. In a fuller model, there would be a range of sites with different breeding productivity, making animals' dispersion decisions more complex.

Table 3
Population Changes in a Source Habitat

Activity	Number after activity	BIDE	Notional season
Census	n_1		fall
Reproduction	$n_1 + \beta_1 n_1$	$b_1 = \beta n$	spring
Death	$n' = P_{A1} n_1 + P_{Ji} \beta_1 n_1$	$d_1 = (1-P_{A1})n + (1-P_{J1})\beta_1 n_1$	winter
Dispersal	n_1	$e_1 = n_1' - n_1$	summer
Note: The notional seasons are out of order for analytical reasons; see text.			

A sink habitat is largely symmetrical, and is shown in Table 4. For example, fall dispersion sees immigration instead of emigration. However, that there is no reason to assume a fixed population n from year to year.. Immigration may be less (or more) than replacement levels, and we should model the number of breeding sites independent of the size of the population. Moreover, the number of dispersers from the source population is determined in the source itself (as $e_1 = n_1' - n_1$) while the immigration into the sink is determined by source dispersal (as $i_2 = e_1$).

Table 4
Population Changes in a Sink Habitat

Activity	Number after activity	BIDE	Notional season
Census	n_2		fall
Reproduction	$n_2 + \beta_2 n_2$	$b_2 = \beta_2 n_2$	spring
Death	$n_2' = P_{A2} n_2 + P_{J2} \beta_2 n_2$	$d_2 = (1-P_{A2})n_2 + (1-P_{J2})\beta_2 n_2$	winter
Immigration	n_2^*	$e_1 = i_2 = n_2^* - n_2'$	summer
Note: The notional seasons are out of order for analytical reasons; see text.			

This last variable, source emigration (e_1), provides the sole link between the two habitats. Managers' utility functions over this emigration, and the effects of policy instruments on this variable, therefore determine whether TBC is feasible. In the model so far, TBC would only make sense if both managers wanted source emigration ($e_1 = i_2$) to change, and both wanted it to change in the same direction.

These conditions are fairly rare, but do occur in the case of an overpopulated ungulate range that lacks natural predators. For example, Rocky Mountain National Park (RMNP) has far too many elk for its range, and reintroducing wolves or other major carnivores is currently not politically feasible. Though regular hunting of elk occurs in Grand Teton NP, and occasional white-tailed deer hunts take place in Fire Island National Seashore and a few other units in the lower 48, hunting in RMNP is not politically possible today. (Elk hunts inside park boundaries have occasionally been tried, beginning in 1941, but they draw too much opposition.) In this setting, RMNP managers share an interest with their neighbors in encouraging elk to move to lower elevations outside the park where they can be shot during the regular hunting season (Buchholz 1983: 182-3).

The other logical possibility occurs if both units would like *smaller* migration between habitats. This too is rare, but bison in the GYE may provide an example (Chadwick 1998). Local ranchers fear that the bison population, which may carry brucellosis, could infect their cattle. Bison hunts are not allowed, except for a newly authorized Nez Perce hunt under terms of the 1851 Treaty of Fort Laramie (McMillion 2006). (Interestingly, elk also carry brucellosis, but because there are legal elk hunts, ranchers do not demand control of Yellowstone elk.) In this political environment, Yellowstone NP managers and their neighbors cooperate to prevent bison dispersal from the park to adjacent national forests and private land.

In addition to source-sink relationships, two habitats might be linked because they are used by animals at different times of the year. My modeling of this migratory connection is non-standard, but again, it will work for our purposes. Suppose there is some mortality rate $(1-P_1)$ that occurs as the animals move from the winter habitat to the summer habitat. Next the surviving population reproduces, and then suffers mortality on the return migration to the winter habitat. The meaning of "emigration" and "immigration" (both modeled from the perspective of the winter habitat) differ here, since they are temporary movements of the entire population instead of permanent movements of a surplus population. For this reason, the model distinguishes this form of emigration with capitalization, as E_1 and I_1 . The model is summarized in Table 5.

Table 5
Population Changes Two Migratory Habitats

Activity	Number after activity	BIDE	Notional season
Census	n_1		winter
Out migration	$n_1' = P_1 n_1$	$E_1 = P_1 n_1$ $d_1 = (1-P_1)n_1$	spring
Reproduction	$n_2 = n_1' + \beta_2 n_2'$	$b_2 = \beta_2 n_2'$	summer
Return	$n_2' = P_{A2} n_1' + P_{J2} \beta_1 n_1'$	$d_2 = (1-P_{A2})n_1' + (1-P_{J2})\beta_2 n_1'$ $I_1 = n_2'$	fall

In contrast to the source-sink model, there are now two variables that matter for TBC. The total outmigration E_1 lies under the control of the winter habitat manager, while the total in-migration I_1 is subject to the summer habitat manager. At the same time, the population E_1 produced by the winter manager becomes a kind of "raw material" for the summer manager, just as the summer "output" I_1 becomes the raw material for the winter range.

As a result, there is significant scope for TBC in a migratory setting, especially if both managers want to increase the number of animals available for their own purposes. There could also be a shared interest if both managers want to reduce the number of animals on the range because of overpopulation, though TBC would be unnecessary since each could simply reduce the numbers on her own. If both wanted to harvest the animals, however, there would be a conflict of interest because animals harvested in one habitat become unavailable to the other. The problem here would be a prisoners' dilemma, and explicit TBC would be helpful. However, if one manger wants to reduce the number of animals while the other manager wish to increase them, this conflict of interest would preclude cooperation.

The first case, in which two managers both want to increase the number of animals, is more common in US wildlife settings. As a result, we would normally expect TBC to be common for these habitats. Even so, the role of TBC will depend on the exact interaction of managers' goals.

A good example of TBC in a migratory setting is elk management in and around Jackson Hole, Wyoming. Elk winter in the National Elk Refuge, managed by the USFWS. The herd disperses widely in the summer, into Grand Teton National Park (GTNP), Bridger-Teton and Targhee NFs, the John D. Rockefeller Parkway, and the southern parts of Yellowstone NP. The USFWS provides supplemental feeding for the elk in winter, which increases the numbers available for tourism in GTNP and for hunting in the Thorofare region in and just south of Yellowstone NP (Ferguson 2003; Righter 1982: 139-140). By statute, GTNP is also mandated to allow hunting by licensed hunters, temporarily deputized as park rangers, when there are surplus elk (as there always are). These agencies, along with the Bureau of Land Management (BLM) and the Wyoming Game and Fish Department, have worked together for decades to manage the herd. In response to bison-related lawsuits, these same agencies have recently proposed new management strategies that continue human intervention in the elk and bison herds while reducing the supplemental feeding (see http://www.fws.gov/bisonandelkplan/index.html).

The greater ease of TBC in a migratory setting is evident in the history of international cooperation to manage wildlife. The first three conservation treaties between the US and Canada all involved migratory animals–migratory birds, fur seals, and (in a more complex case) inland fisheries. The 1916 Migratory Bird Treaty, which is still in effect, kept Americans from shooting birds on the way to Canada, and vice versa. The 1911 North Pacific Fur Seal Convention addressed the problem that seals gave birth on US-held islands but then spent most of the year in international waters, where they were hunted by Canadians, Russians, and later Japanese. The treaty, which included payoffs to countries losing expected seal harvests, lasted until these countries went to war with each other in 1941. A replacement treaty has since managed the issue. The 1908 Inland Fisheries Treaty addressed a variety of species, with mixed effects, but anadromous species such as salmon are managed with mixed success in widespread examples of TBC. The analysis here, focusing on variation in the structure of animal habitats, explains these treaties more parsimoniously than does Dorsey (1998), who emphasizes aesthetics and sentiment, economics, scientific knowledge, and legal precedent.

To summarize, TBC should be common in migratory settings, where each manager controls a variable that is important to the other. In contrast, TBC should be exceptional in populations with source-sink dynamics, limited only to cases in which at least one manager does not (at the margin) value animals on the range. The question remains whether each manager will want more or fewer animals on her range, and I turn to these goals in the next section. After this, I briefly examine the policy instruments that managers use to pursue these goals.

Management mandates and utility functions

The first section argued that reserves' boundaries will be generically too small, and suggested that this problem remains even in an environment of multiple agencies that border one another. As a result, many management objectives will require TBC. After this, I developed a simple model of animal populations across two habitats that showed why TBC should be common for migratory species but depends on particular, and unusual, management goals in a source-sink setting. We must, then, understand management goals and choices.

Agencies might have preferences over both means and ends (see also Freyfogle and Newton 2002). Not intervening in natural processes is a means toward an end (such as natural regulation) but an agency mandate might require that particular means somewhat independent of the goal or outcome of using that policy. This often occurs in national parks, where letting nature run its course produces a very unnatural ecosystem in the absence of natural predators.

Several implications for analyzing management decisions follow from the analysis so far. First, examining *transboundary* cooperation means that we must consider multiple agencies. That differs from the usual modeling strategy of analyzing the political economy of an individual agency (Lueck 2001) or examining how best to achieve a single goal.

Second, examining multiple agencies means that we should consider multiple goals. These agencies might all have the same goals, but the analytically interesting case requires that these agencies will have at least somewhat different goals.

What might these goals be? The usual public choice approach to studying government agencies generally assumes that all agencies have the same goals–namely, the maximization of budget and staff (i.e., Niskanen 1971). This framework has been taken up by much of the new resource economics (i.e., Baden and Leal, eds. 1990).

One natural objection to this approach is that it ignores both what managers think their goals are, as well as ignoring the explicit mandates given to each agency and the legislative oversight that seeks to hold them to these mandates (police patrols and fire alarms). Phrased differently, the USFWS really does think that its job is to manage fish and wildlife, whether in fish hatcheries, preserves for waterfowl or elk, or implementation of the endangered species act (ESA). Clearly the USFWS wants a larger budget and a larger staff with which to manage fish and wildlife, but it does not spend money on digging holes and filling them up again.[3] In addition, Congress clearly assigns different lands to different agencies with varying mandates as a way to create administrative advocates for specific goals in particular regions. Decision rules also provide continuity across time and consistency over space that might not occur with a single agency and multiple mandates.

More abstractly, assuming that all agencies are budget-maximizers tends to assume away the interesting TBC issues. Many TBC problems stem from differences in agency goals–if agencies simply maximize budgets (or staffs), then we have a divide-the-pie game, which is well understood. Moreover, divide-the-pie problems present a problem of pure conflict, while participants identify TBC problems as mixed-motive games combining both common interests and conflicts of interests.

If we take agency mandates seriously, however, we run into another problem. First, most management agencies have complex mandates. The US Forest Service (USFS), for example, has a "multiple use" mandate that includes timber management, grazing, mineral extraction, and recreation. The NPS both preserves natural processes in parks and makes the parks available for the public's use and enjoyment. Parks Canada faces a similar conflict of goals: its foundational legislation in 1930 dedicated its parks "to the people of Canada for their benefit, education and enjoyment and such Parks shall be maintained and made use of so as to leave them unimpaired for the enjoyment of future generations." All these agencies face trade-offs: clear-cutting, oil and gas drilling, and mining all harm recreational values in national forests; public enjoyment may harm natural processes in national parks (Sabatier et al. 1995).

[3]The US Army Corps of Engineers does this, but that is a story for another paper.

Rather than modeling this complexity of real agencies, I will examine only stylized management rules here. For example, an agency might seek to maintain the number of animals of a given species at a particular level. It might seek a particular level of harvest or a sustainable harvest over multiple periods. It might have a "hands-off" policy that allows natural processes to take their own course, as in "natural ecosystem management" (Agee and Johnson, eds. 1988; Grumbine 1994).

For example, an agency might seek to maximize the population of a species, so that $U_{MP} = U(n_1)$, with $\partial U/\partial n_1 > 0$. This has sometimes been the USFWS goal in wildlife refuges, or a NPS goal for charismatic megafauna favored by tourists. Pursuing this goal damages the range and other values, however, and is largely out of favor. If the manager of a source habitat has this goal, however, TBC will generally be impossible because the manager will seek to retain as many animals as possible. The only exception would occur if adjacent managers wanted to get rid of as many animals as possible, which seems unlikely.

More reasonably, an agency might seek to maintain a population in its range that is sustainable over time.[4] For example, the NPS often seeks to preserve complete ecosystems as "vignettes of primitive America" prior to European contact. If we label the population target $\hat{n}_1$, and use Euclidian distance for utility, then agency utility under a sustainable population regime would depend on the difference between the target and actual populations, i.e., $U_{SH} = -(n_1 - \hat{n}_1)^2$. Given the above population model, a target above the actual population ($\hat{n}_1 > n_1$) in a source-sink system will imply that the agency will want to reduce or eliminate emigration, while a target

[4]Such mandates normally consider only one species at a time, ignoring whether, say, a fruit bat population hunted at sustainable levels still provides sufficient pollination services for the ecosystem as a whole (Schaller 2000).

below the actual population ($n_1 > \hat{n}_1$) will mean that the agency will want to encourage greater immigration. The agency might be able to influence migration directly, or might have to manage it indirectly with other policy instruments, discussed in the next section.

If agencies have a population target mandate, TBC will be possible only under if both agencies want to reduce emigration from the source to the sink. This requires that the source agency wants a larger population while the sink wants a smaller one. GYE bison management, though peculiar because of the brucellosis issue and the virtual absence of hunting, represents such as case (see above).

Instead of managing population size, an agency might maximize harvest (h_1). In this case, $U_H = U(h_1)$ with $\partial U/\partial h_1 > 0$ and presumably also $\partial^2 U/\partial h_1^2 < 0$ (diminishing marginal returns). In a one-period model, this implies that the agency will harvest the entire population so that equilibrium $h_1^* = n_1$. This doesn't seem to happen, so some constraint must come into play. I will discuss two possible constraints, sustainability and multiple use.

Fist, the agency could seek to maximize a sustainable hunt (s_1). Instead of modeling this explicitly with multiple periods, we can characterize this as $U_S = U(s_1)$, subject to the constraint $BIDE_1 = 0$, with $\partial U/\partial s_1 > 0$. One result of this agency mandate is that it will result in $e_1 = 0$ since emigrants will be harvested instead.

If this habitat is a source, the incentives to prevent all emigration means that TBC with adjacent managers will be impossible. If the habitat is a sink, however, TBC will focus on increasing immigration into the sustainable-harvest sink. Whether this TBC will occur will therefore depend on the incentives of the source agency. If the source managers are seeking a target population, sink harvesting may provide regular annoyances precluding TBC. One extreme example is the Blackfoot Indian Reservation's policy of advertising trophy-quality hunts for non-tribal members, thanks to the source next door: "The abundant wildlife of Glacier Park wander out onto the mountain front and prairies of the Reservation, making this area prime hunting destination." (Brochure available at http://www.blackfeetnation.com/Home%20Page/F%20&%20W%20Inside%20Brochure.htm, accessed October 2005) The NPS and Blackfoot IR do not cooperate on this issue, as NPS population goals are not facilitated by having a trophy hunt next door.

As opposed to a population target, the agency might have a multiple-use mandate, as does the US Forest Service (USFS). For example, it might seek both a certain level of population and a harvest. In this case, $U_{MU} = h_1 - (n_1 - \hat{n}_1)^2$. Here, the multiple-use agency will reduce hunting if the population lies below the target level, while increasing hunting when the population is above target.[5] Near the target, some hunting will occur because the utility gains from hunting will exceed the utility loss from just missing the target. Whether the population is above or below the target, emigration represents "wasted" animals in terms of the agency's goals, so that potential emigrants will be harvested if the population is over (or nearly over) the target level. With this mandate in a source population, then, TBC will be impossible.

A final agency mandate, natural regulation or non-intervention, also deserves mention. This is not particularly interesting for a single habitat, but may become interesting with two

[5]I should note here that actual multiple-use decisions in the US are complicated by the fact that either the USFS or the USFWS may have the population goal, while the relevant state game agency (or Department of Natural Resources, DNR) maximizes hunting. USFWS monitoring of state plans for formerly-listed species such as wolves and grizzly bears, or coordination between USFS and DNR planners, would yield something like the decision rule here.

habitats because the agency in the second habitat makes decisions knowing that its sister agency next door will do nothing. It is no longer obvious for the first agency that "doing nothing" will achieve the goals of "doing nothing."

With this mandate, TBC will generally not occur since the no-intervention agency does not, by definition, take actions. A good example is grizzly bear management in the Crown of the Continent Ecosystem (COCE) straddling the US and Canada. Though grizzly bears were a listed species in the US until this year, some Canadian regions along the US border have allowed hunting. When it has occurred, Canadian managers have chosen harvest rates that were sustainable for their own population, ignoring the negative effects on the US population (Horejsi 1989; Kansas 2002: 23). Oddly enough, given the scarcity of grizzlies in the US relative to Canada, this lack of TBC reflects the fact that the US population in and around Glacier NP is a regional source; adjacent Canadian regions in Alberta and British Columbia, including Waterton Lakes NP, are population sinks. This means that hunting a "sustainable" population of Canadian grizzlies actually depended on in-migration of "threatened" grizzlies from the US. As the analysis here shows, whether the mandate is natural regulation (in the park sources) or population targets (in non-park US source regions governed by USFWS grizzly recovery plans), TBC with a sink habitat that has a hunting-related mandate will not occur.

As a partial exception to the rule that TBC does not occur under non-intervention mandates, we may see TBC and natural regulation in boundary design matters. In many developing countries, for example, parts of reserves that were once managed for (sustainable) harvest or multiple use may be converted to a non-intervention (and no-hunting) mandate. The effect of changing this mandate will be to increase the population in the new reserve because the previous harvest will not occur. If this reserve is a source habitat, emigration will increase to adjacent habitats. In many of these settings, the management change seeks to create a matrix of reserves, often with a protected source reserve surrounded by sink habitats available for harvest by indigenous peoples (Fimbel et al. 2000; Hart 2000; Hill and Padwe 2000; in the Rockies, see Noss et al. 1996). The analysis here suggests that this management strategy, which we might classify as a form of TBC, can work.

Table 6
Mandates and Source-Sink Habitats: Implications for TBC

Mandate, source	Mandate, sink	TBC
Sustainable harvest	Any	Impossible
Harvest or multiple use	Any	Impossible
Population target	Sustainable harvest	Only if source desires $e_1 > e_1^*$, i.e., $n_1 > \hat{n}_1$
Population target	Population target	Only if both managers desire $e_1 < e_1^*$, i.e., $\hat{n}_1 > n_1$ and $n_2 > \hat{n}_2$
Natural regulation	Any	Impossible (except in boundary design)

Table 6 summarizes the discussion so far, which has been limited to source-sink dynamics. As discussed earlier, TBC will be much less problematic in migratory settings, so that variation in agency mandates yields less variation in outcomes. In source-sink settings, however, one needs to know both the habitat and the mandate. For example, sustainable harvest in a source precludes TBC, while sustainable harvest in a sink may allow it.

Policy instruments

The previous section analyzed how mandates affect TBC in terms of ultimate goals, independent of means. The central variable was e_1, the level of emigration from the source to the sink. This analysis elided the fact that many different policy instruments might affect this emigration level. While we do not yet have a theory of why certain environmental policy instruments are chosen over others (Hahn and Stavins 1992), it is helpful to see how the instruments available map into the BIDE model used here.

In a BIDE model, an agency might allow harvesting (h), which increases the death rate (d). An agency might reintroduce animals into an area (r), which is a form of immigration (i), or transfer animals (t) for relocation elsewhere (e). Births (b) are probably harder to manipulate directly, but supplemental feeding and forage management (f)–both practiced in wildlife refuges in the United States–might add to the birth rate. Finally, an agency might have a hands-off policy that eschews all of these policy instruments. These instruments are summarized in Table 6.

Table 6
Policy Instruments in the BIDE Model

Instrument	Variable	BIDE effect	Each habitat	Effect in source
reintroduction	r	immigration	$(r_1 + i_1)$	more e_1
harvesting/hunting	h	deaths	$(h_1 + d_1)$	less e_1
transfer/relocation	t	emigration	$(t_1 + e_1)$	less e_1
feeding and forage	f	births	$(f_1 + b_1)$	more e_1
natural processes	∅	∅	∅	∅

Because of the importance of emigration out of the source for TBC, Table 6 also shows the indirect effect on e_1 of each instrument. For example, successfully reintroducing wolves into Yellowstone NP led to dispersal out of the original habitat into adjacent sinks. Hunting of any species, by killing would-be surplus animals, reduces e_1. A relocation program that takes swift foxes out of a Wyoming source population and introduces them into a new Canadian habitat will reduce the supply of swift foxes for dispersal into Montana and the Dakotas. Supplemental feeding of elk in the National Elk Refuge increases the number available for dispersing into adjacent national forests.

Each of these policies costs money, which I have not modeled here. However, Table 6 provides a convenient summary showing when a sink manager would wish to commit resources to a policy being carried out in an adjacent source habitat. For example, sink agencies would be willing to commit to reintroduction and feeding programs because these will indirectly increase populations on their own habitats through out-migration from the source.

Conclusions: implications and extensions

This paper has examined the interaction of habitat type with agency mandates to determine under what conditions transboundary cooperation is likely to be successful. While TBC is easiest when managing migratory populations, neighboring agencies managing a population source and a population sink will succeed in cooperating only under rather specific combinations of mandates.

One policy implication concerns managing metapopulations on a regional basis under conditions of landscape heterogeneity. The basic source-sink model in this paper suggests that mixing sustainable harvest in sinks surrounding source reserves managed for target populations or not managed at all (i.e., natural regulation) represents a viable strategy, one with some successes in tropical rainforests (Robinson and Bennett 2000). However, this strategy does not imply unconditional acceptance of the wildlife goals of indigenous peoples surrounding the source, as is often recommended in the name of "community participation" (see *inter alia* Bawa et al. 2004; for a critique, see Agrawal and Gibson 1999).

This strategy can be extended to mixed public-private management, as is being tried with the jaguar population on the Arizona-Mexico border. Defenders of Wildlife, Naturalia, and other wildlife groups have created a Northern Jaguar Project seeks to own 40,000 acres of source habitat. At the same time, it provides monetary incentives for ranchers in adjacent sink habitats not to kill jaguars. For example, a bounty program will pay ranchers as much as $500 for each photograph of a jaguar on his or her land (Friederici 2006). Again, the analysis here suggests that this is a viable strategy.

At the same time, the analysis here does not support other types of regional strategies. For example, Agee and Johnson (1988b) argue that regional metapopulations can be managed by multiple uses at the regional scale, implemented by single-use mandates by individual management units within the region. We have seen that TBC among single-goal agencies is very problematic for most combinations of habitat and mandate. If TBC proved impossible, this would prevent agencies from reaching the multiple goals at the regional level if there were any externalities across the units.

Of course, the analysis here is admittedly incomplete. One heroic assumption is that managers have complete information about the workings of the ecosystem. I have also ignored ecosystem effects of various kinds. For example, managers have historically eliminated predators so as to maximize the ungulate population (Bennett and Robinson 2000). This remains policy in Alaska, where 1994 legislation provides for "high levels of harvest for human use." This harvest focuses on maximizing moose, caribou, and deer, which means that wolves and other predators threaten that goal (Jans 2006). Idaho sought a similar strategy, subject to USFWS veto because its wolf population was (until recently) listed as threatened (see Nie 2004: 214). One manager's policy toward predators therefore has important effects not only on that species but on that species' prey animals in adjacent habitats, complicating TBC considerably.

In addition, I have focused on agencies with wildlife-related goals only. One natural extension would combine wildlife and non-wildlife goals. Dams and water conservation, pest control, agricultural subsidies and resource extraction all affect wildlife; Randy Simmons (2002: Chapter 2) claims that such government actions threaten 58% of all listed species.

Another possible direction for research would extend this approach to other land management problems with externalities. For example, the NPS, USFS, and suburban areas have very different fire management regimes, yet wildfires can easily spread from one parcel to another. My intuition is that some combinations of fire management and terrain lend themselves to interagency cooperation, while other combinations do not.

The approach here could also be extended to other resource management issues. Like wildlife management units, these do not normally coincide with natural boundaries. For example, freshwater is best managed at the level of the intact basin, but boundaries rarely follow watershed lines. Again, different units have different goals, whether flood control (Army Corps of Engineers), agricultural development (Bureau of Reclamation), instream flow (USFWS) or clean water (EPA). Parcels are asymmetric, since some lie upstream of others. This mixture of terrain and mandate should also yield variation between successful and failed TBC (see Dana 1990). Analysis would also help us understand better regional problems such as freshwater flow through the Everglades ecosystem.

Such problems of transboundary cooperation are ubiquitous, and found in a wide range of environmental problems. Taking land-use mandates seriously, while looking at variation in the nature of the underlying resource, has given this paper a framework with which to explain variation in the success of such TBC from one setting to the next. Rather than being a matter of administrative or political "will," as often portrayed, TBC can be understood in more mundane ways through these mandates.

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