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Gamebird 2006

A Joint Conference - Quail VI and Perdix XII
Managing Gamebirds in the 21st Century

Warnell School of Forestry and Natural Resources
The University of Georgia, Athens, Georgia
31 May - 4 June, 2006

Gamebird 2006

A Joint Conference of Quail VI and Perdix XII
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Foreword

In his concluding remarks in Quail IV, John Roseberry stated,"the ultimate challenge for quail conservation was to change how society managed its forests and fields if bobwhites are to remain a widely hunted game species". He predicted that with on-going research we will have the knowledge to produce locally abundant quail populations, even record numbers, but recovering quail at a landscape scale is a daunting, perhaps impossible, dilemma for researchers, managers, and policy makers. At this meeting, Dick Potts lamented on the decline of Grey Partridge in Europe, from millions to tens of thousands, even though intensive management can demonstrate restoration to "Edwardian" numbers. The continents and species may differ, but the tune is the same. The challenge for the recovery of these flagship species, not to mention the hundreds of less prestigious Galliformes, is what biologists around the world face and rendered an excellent rationale for hosting a National Quail Symposium and Perdix Conference simultaneously, now 3 years ago.

I met John Carroll and his family on my first visit to the Game Conservancy Trust in 1997 just as he was leaving to head to the University of Georgia to start a game bird program. It was a trip that my boss, at the time Lenny Brennan, supported and encouraged. Since then we have collaborated on dozens of research projects. This is to say, that while game bird management is a small world, one purpose for combining these symposia was to make it a tad larger for all attending. In this volume there are presentations from 8 countries and 3 continents. From the EU, information on conservation headlands to biodiversity plans and from the US farm and forest management to NBCI are presented as shared visions for conservation. There is important information on the efficacy of management techniques, and interestingly re-introduction and translocation practices are tested, which is likely to become an important conservation practice for some species. As a game bird biologist, it is more than just a novelty to learn about how others approach management of their wildlife resources, it is another window into what drives game bird populations which makes us all better biologists. One of the highlights of the meeting itself was the panel presentations and discussion on the effects of radio-transmitters on quail as it is vital that our methodology remains as unbiased as possible.

The editors of this symposium deserve credit for their breadth of knowledge to review and edit manuscripts from species around the world. They have done an outstanding job elevating the quality of the science for a span of disciplines. Probably one of the greatest testaments to the resurgence in research on game birds is the number of bright and ambitious graduate students attending, and in this case, running the meeting as well. Certainly, they do the lion's share of the work in developing game bird knowledge and in this case pulling together the symposium for publication. I toast their dedication and thank all the presenters that took the time to make Gamebird 2006 a success. With our collective efforts perhaps the challenges outlined by Roseberry and others before him can be met.

Bill Palmer, Ph.D.
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Acknowledgements

Gamebird 2006 came together because many of us believe that the flow of information must not be hindered by political boundaries. Like any endeavor, there are many contributors to these proceedings. Those who funded the conference, especially the organizers of Quail V, who provided seed money, Quail Unlimited (QU) National and the Northeast Georgia Chapter of QU are especially thanked. We have listed all of the conference sponsors on the following pages.

During the conference we thank the four plenary speakers for their thought provoking and diverse presentations. Like always, session chairs made sure that everything went smoothly. We acknowledge all the authors for their diligence and patience while we put this together. We specially thank the reviewers who often contribute a great deal of work but are often overlooked. Again, all of the reviewers are listed on the following pages. It took longer than we expected and I take responsibility for that delay (JPC). We thank JP Bond for his graphics expertise and Audrey Owens for her artwork. We thank Bill Palmer for writing the Foreword and to the Tall Timbers team for always being around to help out.

Finally, we thank Dick Porterfield, former Dean, D. B. Warnell School of Forestry and Natural Resources, who provided manpower and logistical and financial support for both the conference and these proceedings.

We hope that this one time merging of the two conference series will help gamebird management by creating a little bit smaller world for those interested in the conservation of these great birds.

Cheers and good reading,
The Editors

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Invited Manuscripts

Integrating Management, Research, and Monitoring: Balancing the 3-Legged Stool

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Research and monitoring programs are often thought of as competing with "on the ground management" for attention and funding. This is false trichotomy; instead, it is more appropriate to view management, research, and monitoring as complementary endeavors, in which loss of any 1 of the 3 is disruptive to the remaining 2. There is often significant or even profound uncertainty about the system's likely response to management, beyond environmental and other sources of uncontrolled variation. Sometimes this uncertainty can be reduced through directed research studies, including experimentation. However, management decisions usually cannot await the completion of elaborate, multiple-year studies. Adaptive resource management (ARM) provides managers a way to make optimal decisions with respect to resource objectives, given the current level of uncertainty about system response, *and in anticipation that learning will improve decision-making through time*. Under ARM, resource goals and objectives are *always* paramount and research and monitoring programs exist to provide managers with the tools they need to make better decisions. The essentials of ARM are clear, compelling, and critically needed in natural resource management. We can no longer afford the luxury, if we ever could, of management divorced from research and monitoring, and vice versa. By keeping the focus on management decision-making and resource objective outcomes, ARM places an explicit value on research and monitoring that then can be used to justify monitoring and research programs.

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Key words: adaptive resource management, decision making, research and monitoring

Introduction

In our experience in working with natural resource managers and researches, we often encounter situations where management, research, and monitoring activities are viewed distinctly. Management is typically viewed as involving the concrete, hands-on, practical aspects of conservation: preserving and managing habitats, regulating harvest and trade, and other aspects of "on-the-ground" work. Research, although recognized as important, is often viewed as less important than monitoring, and certainly than management-somewhat as a luxury of academia that we should do, but only if we have sufficient time and funding left. Monitoring is viewed as a way of assessing the status of populations, communities, and ecosystems, but typically is not formally connected to conservation decisions.

Here we argue that management, research, and monitoring are actually complementary, not competitive activities, all 3 are important to successful conservation, and loss of any 1 of the 3 disrupts the other 2. We use the metaphor of a 3-legged stool to convey these ideas.

Management As Modeling

Management is simply taking an action to obtain some desired resource outcome. It requires a range of alternative actions that can be taken, and specification of an objective that we are trying to achieve. Examples of management include: the application of prescribed fire to increase or improve habitats and, presumably, sustain larger populations; the setting of harvest regulations to provide recreation, control populations that may be damaging habitats or otherwise causing problems, and to providing eco-

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nomic benefits; the construction of reserves to maintain species diversity, protect endemics, and/or provide corridors of movement among discrete habitats or populations.

Research is a process of inquiry that includes description of natural systems, but also involves addressing questions about how these systems function. Thus, research would include testing and quantifying ecosystem feedback relationships and mechanisms of population regulation, to name two. *Monitoring* involves the observation of natural systems through space and over time, and may be descriptive (i.e., simply oriented toward quantifying patterns or trends), but may also be connected directly to research (by providing answers to testable predictions) or management (by providing feedback about the results of management actions).

As we discuss below, we view management, research, and monitoring as highly complementary activities, whose boundaries are often blurry. However, there are unifying ideas, and one of these is the idea of a model. Conservation managers usually do not think of themselves as modelers. However, even if managers are not conscious of the fact, every management action involves a decision that is made to reach a goal, and at least implicitly involves a model. For example, a manager may desire to increase carrying capacity via habitat modification such as prescribed fire. Implicitly, he or she believes that certain actions (e.g., a burn) are likely to have the desired results (habitat improvement), and these outcomes are more desirable (have higher objective value; Figure 1a). This belief is a conceptual model of how the system is likely to respond to management, whether or not it is formalized into a mathematical model.

Uncertainty In Management

Sources Of Uncertainty

The reality is that uncertainty nearly always confounds a simple decision model (such as Figure 1). That is, the manager can never be sure with 100% certainty that any given decision will result in the desired outcome. Management uncertainty comes in 4 basic types: environmental uncertainty, partial controllability, partial observability, and structural uncertainty; we emphasize the last.

One basic but important form of uncertainty is that due to the fact that habitat and populations are influenced by factors that may not be under management control. For example, if we decide to burn a woodland to improve habitat conditions, a disease outbreak or unusually severe winter may occur that results in a lower than predicted population response. Likewise, even if we don't burn, other favorable factors may cause the population to perform better than predicted. The influence of factors in the environment that are unpredictable, and that add to the influence of our management decisions, is termed *environmental uncertainty*. A similar result can occur because the management itself is only partially controllable, for instance, a burn may be cooler or less extensive than planned, resulting in a poor response by the population. This is referred to as *partial controllability* (Figure 1b).

In addition to these 'real' sources of uncertainty, monitoring programs generally will not be able to perfectly measure the systems response to our management. Especially when we are monitoring abundance and other population or community attributes, these will usually be based on some type of statistical sample, and thus subject to error. This is referred to as *partial observability*, or sometimes, statistical uncertainty (Figure 1c).

Finally, in addition to all the above sources of uncertainty, we return to an idea we started with, namely that management implicitly involves acting under a model of how our system is likely to respond to management. This model contains, at least implicitly, current knowledge as to how the system functions, which is presumably based on past observation and research. However, this past knowledge basis is seldom unequivocal, and is often very incomplete. Unless we are absolutely certain about the basic mechanisms that determine our system, we should be honest and admit that this model is but one hypothesis about how the system works, and that it may not be the best model. In the prescribed fire example, model 1 is that burning provides a ben-

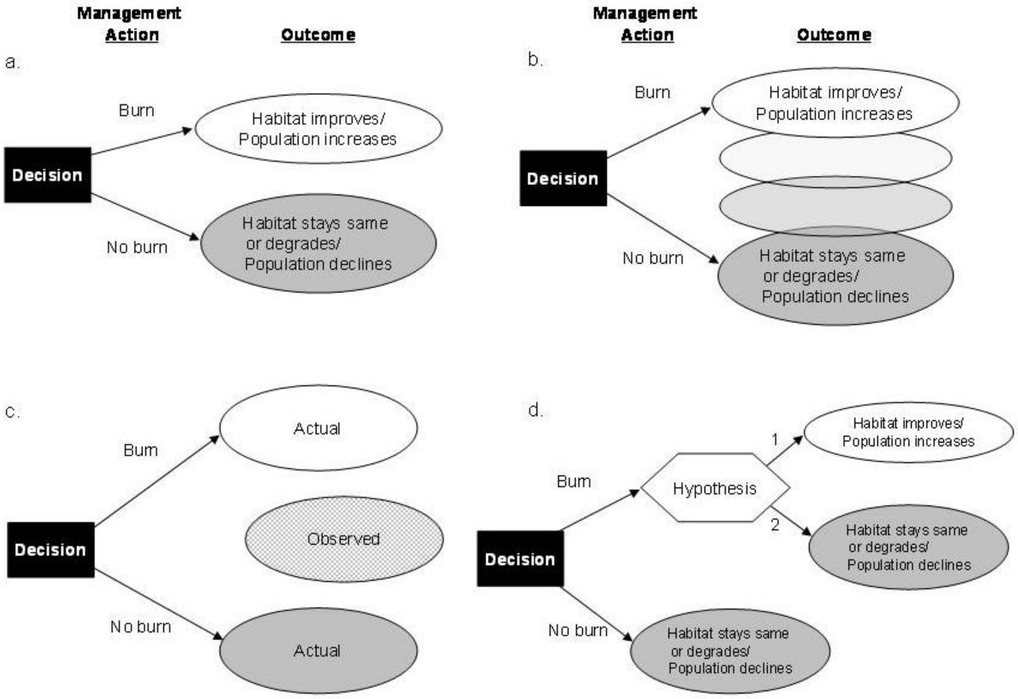

Figure 1: Schematic of a hypothetical decision model for effects of prescribed fire on habitat and population response with (a) no uncertainty in decision-outcome, (b) environmental uncertainty, (c) partial observability, and (d) structural uncertainty. Here objective values could include the number of hectares of good habitat and/or population size.

eficial impact, but we should at least consider the possibility that model 2 (no discernible impact) is correct. We refer to this last source of uncertainty as *structural uncertainty*. Seen in this light, structural uncertainty is both a research issue-it occurs because our system understanding is imperfect-and a management issue-resolving or reducing it leads to better decision making.

Dealing With Uncertainty In Decision-making

We begin with the recognition that, although there are several possible ways of dealing with uncertainty, ignoring uncertainty can have severe consequences. Failing to deal with uncertainty may lead to a false sense of security in decision-making and ultimately compromises our ability to reach our conservation objectives. We favor the approach of incorporating uncertainty into the objective by means of *expected values*. Expected values are simply a form of weighted averaging, in which objective values under different possible decision-outcomes are weighted according to the probability that each outcome occurs. The decision-maker than selects the decision that results in the best (e.g., maximum) objective value on *average*, which is the expected value. One important implication of this approach is that reducing uncertainty– if it can be done– has measurable value in terms of the conservation objective. In fact, it is possible to calculate how much improvement could be made in decision-making, were it possible to completely eliminate uncertainty; this is known as the *expected value of perfect information* (Lindley 1985, Clemen 1996) in decision-making.

Reducing Structural Uncertainty

Some types of uncertainty, such as environmental uncertainty, are essentially impossible to control. These must be considered in decision-making, but in all likelihood cannot be reduced (unless we consider artificially controlling the range of environmental variation, e.g., via water control devices). Others can be at least partially reduced by concerted effort: e.g., better field techniques may reduce (but likely not eliminate) partial controllability and better survey methods may reduce partial observability.

We devote special attention to structural uncertainty, because it is the one source of uncertainty that 1) is very frequently ignored, and 2) can be reduced through time via an adaptive approach. Before discussing adaptive approaches, we mention the two other major approaches that can be used to reduce structural uncertainty, because readers are likely more familiar with these approaches, they have occurred more frequently in the literature, and they continue to have merit.

Experiments - which we define as involving control, randomization, and replication of independent subjects - are the "gold standard" of scientific inquiry. Experiments clearly are ideally capable of reducing uncertainty very quickly, and thus are attractive. However, realistic experiments at any meaningful spatial scale are difficult or impossible to conduct in most conservation systems. In addition, because experiments are directed at scientific hypotheses, rather than management objectives, they are not necessarily efficient means of reducing uncertainty for decision-making.

In contrast to experiments, *retrospective studies* are based on an examination of patterns in data that have been collected in the past; thus they are analyzed "retrospectively." These often can provide a good initial basis for the construction of alternative hypotheses and predictive models used in conservation. However, potential explanatory relationships are actually correlative, because of the lack of controls, and are typically confounded with other factors. As a typical example, Conroy et al. (2002) retrospectively investigated the potential influence of habitat, hunting, and competition with mallards (*Anas rubribes*) on populations of American black ducks (*Anas rubripes*). They detected evidence for the impacts of all three factors, but could not infer causation because of confounding (e.g., habitat declined and mallard competition increased over the same period). Conroy et al. (2002) were able to construct predictive models, but other approaches such as experimentation (Anderson et al. 1987) or adaptive resource management (below) are required to reduce structural uncertainty for this problem.

Without denying the importance of both experimentation and retrospective analysis, we advocate a third approach, called adaptive resource management (ARM; Walters 1986), as being generally more suited to conservation decision-making. We especially like the ARM approach because it fits nicely with the idea of multiple working hypotheses (Chamberlin 1897), which we advocate instead of null hypothesis testing. ARM can be implemented in virtually any resource system, and has the advantage of being directed at meeting the conservation objective, not at meeting a scientific objective per se. In fact as we will elaborate below, conducted properly, ARM involves no tradeoff whatever in meeting the resource objective, and thus would appear to be the optimal means of incorporating information into decision-making and reducing uncertainty. Below we lay out the principal elements of ARM, provide some simple examples, and address some common myths and misunderstandings that have contributed to the (so far) relatively rare use of ARM in practical conservation.

Elements Of Adaptive Management

ARM consists of 3 essential components. The first is explicit predictions of the effect of management actions on resource objectives (e.g., population size, harvest) under 2 or more models. These provide the means for comparing the relative support for different management actions. Here, structural uncertainty is expressed in the form of alternative models (e.g., hypotheses) of system dynamics (Fig-

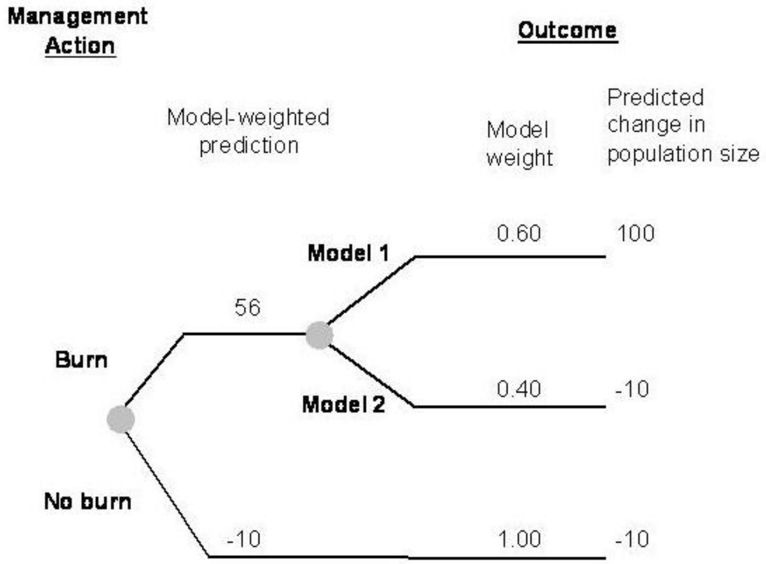

Figure 2: Schematic of a hypothetical decision model of the effects of prescribed fire decision on population response with 2 alternative models. Model 1 predicts an increase in population of 100 in response to burning, whereas model 2 predicts a decrease of 10. Using the model weights shown, the model averaged prediction of population response to burning would be an increase of: 100×0.6 + -10×0.4 = 56.

ure 1d). The set of models should be only as large as is necessary to include the biologically plausible representations of system dynamics. During each decision opportunity, predictions are made under each alternative model, weighted by the relative support for the model, and combined across models (Figure 2). Decisions then are made based on comparing the model-averaged predictions associated with each management action. Although model weights change as information is accrued (more below), the assignment of initial model weights is relatively flexible and can be based on retrospective analyses, expert judgment, or assigned equally among models.

Sequential decision-making is another requirement of ARM and is frequently encountered in natural resource management. Sequential decision-making involves tracking a resource (e.g., population, habitat condition) through time and making decisions based, in part, on the observed status of the resource (Figure 3). The set of management objectives and actions are usually constant, so that the same (or similar) decisions are continuously revisited. Sequential decision-making need not take place on an annual basis and can occur in space as well as in time (Figure 3). The former is particularly useful in situations where decisions will not be revisited at a particular site on a short time horizon but are made over a number of sites. Information feedback, in this sense, is used to improve future decisions at sites that have yet to be managed. Regardless of whether sequential decision-making is through space or time, the key is to provide feedback on the effects of management actions in a timely manner to improve future decision-making.

Monitoring is the third required component of ARM. It provides the information that is used to resolve the key uncertainties - chiefly, structural uncertainty. As described above, structural uncertainty is expressed quantitatively as model weights or relative evidences supporting each model, which can be viewed as probabilities that each respective hypothesis best represents “truth”. To resolve this uncertainty, we need to determine which model best approximates the system dynamics and update the

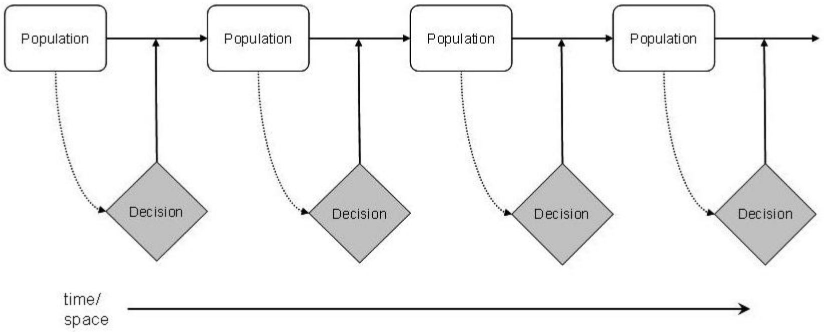

Figure 3: A graphic representation of a sequential decision-making process, through time or space, with population as the resource of interest.

weights to reflect our newfound knowledge. Operationally, this is accomplished by comparing model predictions to subsequent observations of the status of the resource (e.g., population size). Thus at a minimum, monitoring must include a measure of the status of the resource that is consistent with prediction (e.g., if population size is predicted, population size must be measured). The prediction that more closely matches the observed status results in a higher likelihood value and a corresponding increase in the weight for that model. This new weight then is used to estimate the model-averaged predictions for comparing alternative actions at the next decision time. Thus, prediction, management, and monitoring are all connected in a closed loop (Figure 4). In addition to structural uncertainty, the additional sources of uncertainty due to partial observability, partial controllability, and environmental uncertainty, must also be accounted for in the decision model. The general approach is to use probability modeling to account for these factors, either implicitly or explicitly. This is important, both because it gives a more honest picture of the rates of learning under ARM, and helps to direct research and monitoring priorities to reducing uncertainty, where feasible.

ARM: Myths And Misunderstanding

Although ARM appears to be a useful approach to managing gamebirds, to our knowledge, ARM has only been formally applied to waterfowl harvest decision-making (Johnson and Williams 1999). The failure to implement ARM is may be due to institutional resistance (Samson and Knopf 2001), but we think it is also attributable to widespread misconceptions concerning the nature of ARM. Perhaps the most common misunderstanding is that ARM is research. ARM is first and foremost *management*. The primary objective of ARM is to make the best decision with respect to management objectives. Learning occurs as a byproduct of management rather than experimentation. In fact, experimentation (a.k.a. probing the system) can be suboptimal because the system can be driven to a state that is undesirable, potentially reducing future returns (Williams et al. 2002). For example, experimental burning may cause the system to revert to a vegetational community that does not support gamebird populations. In ARM, the goal of learning is to reduce the uncertainty that has the greatest direct impact on decision-making. Thus, learning is targeted on those key components that result in improved decision-making and presumably, greater resource gains.

Another common ARM myth is that it is too risky. We contend that natural resource decision-

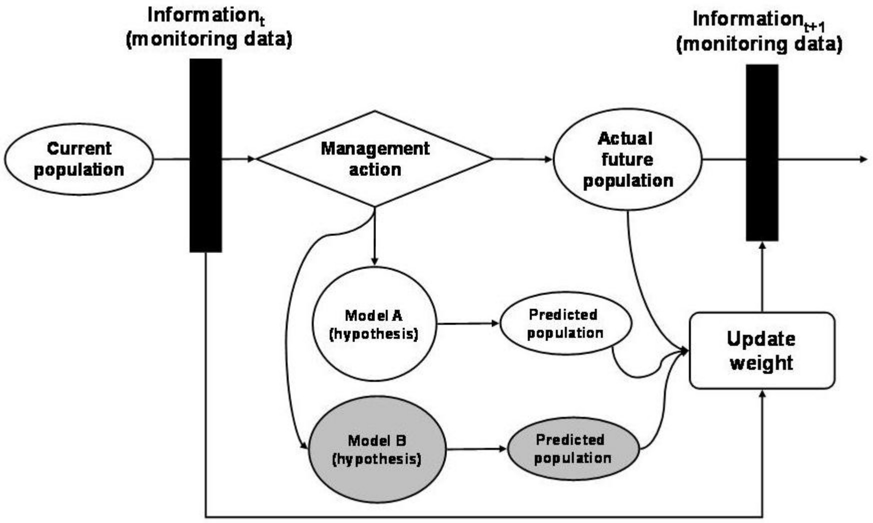

Figure 4: Components of adaptive resource management include: prediction under alternative structural models; feedback of monitoring information to updating weight on alternative models; and adaptive updating through space or time.

making is inherently risky. As we discussed earlier, decision-making is fraught with uncertainty. Hence, all management actions (or inactions) can have unintended and unanticipated consequences. Uncertainty can be reduced by the acquisition of greater knowledge through study and experimentation, which can take considerable time and as discussed above, can force a system into an undesirable state. Management decisions, however, often cannot be delayed until sufficient knowledge has been acquired. Given that decisions under greater uncertainty are riskier than those under less uncertainty, procedures that reduce uncertainty also reduce risk. ARM reduces uncertainty through management and thereby reduces risk. Further, ARM is always directed at achieving the resource goals. Thus, any reduction in uncertainty is not at the expense of, but in addition to, resource gains.

Beliefs that ARM is costly and complicated also are unfounded. Most agencies currently perform most of the tasks required for ARM and hence, ARM would not require additional expenditures. For example, choosing and implementing management actions, monitoring, and sometimes modeling expected outcomes are common practices. All that is then required is a formal means of integrating these components. This integration does not need to be complicated and can be completed with available user-friendly software, such as Netica (Norsys Software Corp., Vancouver, BC, Canada). In fact, the use of simple (but useful) models is preferable in ARM (Williams et al. 2002). Additionally, the evaluation of the sources of uncertainty during the ARM model development is useful for prioritizing and focusing monitoring efforts on only those factors that matter, which can translate into greater cost efficiency.

Unbalancing The Stool

In an era of shrinking budgets and increasing expenses, managers are often faced with decisions on how to cut costs. Unfortunately, one common response is to eliminate what are believed nonessential programmatic elements. We argue that management, research, and monitoring are *all* crucial for natural resource conservation and that the loss on any one of these elements reduces the effectiveness of the others. The elimination of research often results in stagnation, where new scientific hypotheses/ ideas do not become part of management. This also perpetuates a false separation of "management" from "science," thereby reducing the effectiveness of the former and eliminating the context for the latter. Similarly, the elimination of monitoring reduces the effectiveness of management because decision makers no longer have a basis for judging how system is performing in relation to management objectives. Without the feedback provided by monitoring, there is no ability to assess model predictions with data, which eliminates the potential for learning about how systems operate. By contrast, the example of ARM for the management of North American waterfowl exemplifies how management, research, and monitoring can be integrated to form effective, scientifically based decision-making.

When active management is eliminated, decisions are then made by default rather than directed toward an objective. In this context, research and monitoring programs no longer have explicit value. Learning may still occur passively if monitoring continues, but progress would be considerably slower. However, if monitoring also is eliminated (e.g., if animals are no longer harvested, then tag recoveries will no longer be available for survival estimation), learning is prevented.

Summary

Management, research, and monitoring programs are appropriately viewed as mutually supportive of conservation goals, where the loss of any 1 of the 3 is disruptive to the remaining 2. Management explicitly includes the goals of the decision maker and other stakeholders in evaluating the possible consequences of any potential action. Research allows us to state the possible consequences of management actions as predictions, which can be then be used to compare alternatives and select one that leads to a decision that appears most likely (taking into account uncertainty) to achieve our goals. Monitoring provides us with information about the state of the resource system, so we can judge whether we are approaching or diverging from our stated goals, as well as information feedback that allows us to test the predictions of our decision models, and reduce uncertainty through time. This "closed loop" process, known as ARM, formally integrates management, research, and monitoring for more effective natural resource decision-making. ARM provides a mechanism for dealing with uncertainty - inevitable in conservation decision-making - while always keeping resource goals and objectives as paramount. Under ARM, research and monitoring programs have explicit value in terms of the resource objectives are clear, compelling, and critically needed in natural resource management.

We view all 3 of these legs - management, research, and monitoring - as essential to sound conservation. Removal of any 1 of the legs is disruptive to conservation, and ultimately counterproductive. In particular, action-oriented management is sometimes pitted against research and monitoring in the competition for limited funds. This sets up a false choice, a bit like asking whether children need food or education in order to become productive adults. In contrast, under ARM, research and monitoring have explicit value for their contributions to decision-making. Conversely, we "learn by doing," with management actions providing the grist for the testing of critical assumptions, ultimately reducing uncertainty and improving decision-making.

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Management of Southern African Gamebirds: Opportunities and Threats

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Three evolutionarily quite distinct groups of galliforms contribute to a healthy wingshooting industry in southern Africa: guineafowl (*Numida* spp.), francolins (*Scleroptila* spp.) and spurfowls (*Pternistis* spp.). Some species, such as the helmeted guineafowl (*N. meleagris*), Swainson's spurfowl (*P. swainsonii*) and greywing francolin (*S. africanus*), thrive in moderate to heavily disturbed landscapes, mainly agriculture. In fact, helmeted guineafowl and Swainson's spurfowl increased both in abundance and range during the 20th century. Others, such as the redwing (*S. levaillantii*) and Orange River francolins (*S. levaillantoides*) are very sensitive to certain types of land use. These strikingly different responses to land use require equally diverse strategies in order to develop truly sustainable management strategies and policies. These are discussed in detail for each of these five species of gamebirds.

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Introduction

Three phylogenetically distinct groups of galliforms contribute to a healthy shooting industry in southern Africa: guineafowl (*Numididae*), francolins (*Scleroptila* spp.) and spurfowls (*Pternistis* spp.). Some species, e.g. greywing francolin (*S. africanus*), helmeted guineafowl (*Numida meleagris*), Swainson's and cape spurfowl (*P. swainsonii* and *capensis*) thrive in moderately to heavily human-modified landscapes. Helmeted guineafowl and Swainson's spurfowl actually increased both in numbers and range during the first two thirds of the 20th century. Others, e.g. redwing (*S. levaillantii*) and Orange River francolins (*S. levaillantoides*), are very sensitive to certain types of land use. These strikingly different responses to land use require the application of equally diverse management strategies to maintain populations for hunting. These are discussed in brief for each of these five gamebirds.

Here I summarize key points emanating from the results of more than 50 person-years of research on southern African gamebirds by myself and my students: Rob Little (greywing francolin and cape spurfowl), Ray Jansen (redwing francolin and Swainson's spurfowl), and Lionel Pero, Luthando Maphasa, Gerard Malan, Charles Ratcliffe, Helen Prinsloo and Ian little (helmeted guineafowl). Much of what I write below is summarized in Little and Crowe (2000) and discussed in detail in the papers listed in the bibliography.

Helmeted Guineafowl

The helmeted guineafowl may use the frequency of heavy rainfall as a cue to initiate breeding, given that year-to-year variation in their populations (and bags) are positively correlated with monthly frequency of rainfall (Crowe and Siegfried 1978). The availability of arthropod food (critical for successful breeding) is also strongly correlated with rainfall. However, population increases were lower following successive years with high rainfall, perhaps suggesting that the population is limited in a density-dependent manner (Crowe 1978).

Starting in the mid-1980s there were reports of wide-scale collapses of guineafowl populations outside of protected areas. I originally attributed these

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to several successive years with lower than normal rainfall (Crowe and Siegfried 1978), but I was wrong.

Subsequent research showed that the number of pesticides used, individual pesticide toxicity levels and percentage of land under cultivation are negatively correlated with guineafowl populations. However, Swainson's spurfowl populations are positively correlated with these same factors (Pero and Crowe 1996). The fundamental cause of guineafowl population declines and collapses in agricultural landscapes was population fragmentation leading to the undermining of meta-population structure (Ratcliffe and Crowe 2001*a*). In other words, when local fragmented subpopulations came under threat (for whatever reason) they could not be resuscitated by immigration from adjacent subpopulations.

Guineafowl in human-transformed landscapes do not have a home range as traditionally understood. Their daily and seasonal movements are determined by the dispersion of habitat focal points: roosts, drinking water, and habitat for feeding, dusting, and cover (Prinsloo 2003). Indeed, at a revegetated coal mine where all focal habitat components were closely clustered, guineafowl flocks hardly moved during the day (Little et al. 2005). In agricultural areas they also prefer a mosaic of habitats especially with large amounts of edge in proportion to area (Ratcliffe and Crowe 2001*a*,*b*,*c*). So, the critical management strategy for the management of guineafowl as a game quarry species is to cluster habitat focal points and thus maintain multiple local subpopulations in restricted areas.

Orange River Francolin

Unlike that of helmeted guineafowl, the Orange River francolin population in an arid grassland within a protected area collapsed to virtually nil after a five-year bout of overgrazing during the 1930s (Berry and Crowe 1985). This still very poorly studied francolin is one of the most sensitive birds to the grazing and burning of its habitat. Only pristine (rarely burned and ungrazed) arid grasslands can support shootable populations and then at very low levels (Berry and Crowe 1985).

Redwing Francolin

A habitat gradient analysis showed that the abundance of the redwing francolin and other grassland birds in highland wet grasslands is negatively correlated with both grazing pressure and the frequency of burning (Jansen et al. 1999, 2000, 2001*a*,*b*). Once again, as with Orange River francolin, it is absolutely essential to minimize the negative effects of grazing and burning (i.e., removal of cover and damage to leguminous food plants) on redwing habitat if a shootable surplus is to be provided.

Greywing Francolin

Unlike the previous two francolins, the greywing francolin thrives under moderate sheep grazing (Little and Crowe 2000) and can even withstand as much as a 50% annual shooting offtake (Little and Crowe 1993*a*,*b*,*c*). For this reason, this francolin is considered as one of southern Africa's champagne gamebirds, earning shoot operators as much as five times the fees paid for other local gamebirds.

Cape Spurfowl

Like the helmeted guineafowl, the cape spurfowl thrives in golf courses with large amounts of edge habitat and the rough dominated by alien trees and brush (Little and Crowe 1994). This spurfowl certainly benefits from habitat transformation by humans through its ability to colonize alien vegetation and suburban parklands. However in agricultural land it requires patches of fragmented, but closely situated, natural (mainly Mediterranean) vegetation for successful roosting and nesting. For these reasons, I believe it is currently underexploited as a gamebird.

Swainson's Spurfowl

Unlike all other gamebirds discussed above, this species has, in most instances, benefited enormously from agriculture; primarily due to the increased food availability found in growing crops, especially maize (Jansen and Crowe 2002). However, Swainson's spurfowl populations have shown declines if its preferred breeding habitat (islands of brush and trees) is eliminated (Jansen and Crowe 2002). Because of its general prevailence, Swainson's spurfowl is often the focus of what are locally known as community shoots. Some of these community shoots involve 100 or more hunters and may ultimately threaten local populations because of the massive, short term (i.e. over a weekend) offtakes (Jansen, unpublublished data).

Summary

Each southern African gamebird discussed above has its own management 'signature' and thus gamebird management is situation dependent, e.g. in protected areas vs human-transformed landscapes. Moreover, like the much more intensively studied grey partridge *Perdix perdix* (Potts 1986), before we can provide an ironclad management strategy for any of these species, situation-specific research is still a necessity.

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Taking the Northern Bobwhite Conservation Initiative to the Next Level

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During the last few decades of the 20th century, changing and intensifying human uses of land converted and rendered unsuitable hundreds of millions of acres of northern bobwhite (*Colinus virginianus*) habitats, resulting in a long-term population decline of 3.8% per year across the Southeast. During that period, bobwhite conservation efforts were largely ineffectual. Following the success of other national bird conservation initiatives, the Southeastern Association of Fish and Wildlife Agencies (SEAFWA) in 1998 charged its technical arm, the Southeast Quail Study Group (SEQSG) to develop a regional, habitat-based bobwhite recovery plan with population goals and habitat objectives. The Northern Bobwhite Conservation Initiative (NBCI) was published in March 2002, as the first-ever ecosystem-based regional management plan for a resident game bird in the U.S. The NBCI catalyzed immediate major successes in conservation policy, priority, energy and actions, such as the new "Habitat Buffers for Upland Birds" practice in the Conservation Reserve Program. The NBCI also stimulated unprecedented unity of purpose and collective will across the bobwhite community, under the leadership of the SEQSG. A thorough revision of the NBCI already is in progress, coordinated by Tall Timbers Research Station.

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Introduction

The job of stabilizing and restoring northern bobwhite (*Colinus virginianus*) populations across most of the species' range is major unfinished business of the wildlife management profession. Wildlife conservationists already have succeeded in stabilizing and restoring numerous species of wildlife whose populations once were severely diminished. For example, white-tailed deer (*Odocoileus virginianus*), wild turkey (*Meleagris gallopavo*), American alligator (*Alligator mississippiensis*), bald eagle (*Haliaeetus leucocephalus*), peregrine falcon (*Falco peregrinus*), giant Canada geese (*Branta canadensis maxima*), river otter (*Lutra canadensis*), and wood duck (*Aix sponsa*) populations have been restored to secure or even abundant levels. Populations of bobwhites and several species of grassland and early successional migratory birds, however, are experiencing serious long-term declines across much of the U.S.A. Here I provide an overview of the Northern Bobwhite Conservation Initiative (NBCI Dimmick et al. 2002), where we were, where we are, and where we are going.

Where Were We?

Bobwhites are one of the most researched wildlife species in North America. Thousands of scientific papers and several major books have been published about the species over more than a half-century, including *The Bobwhite Quail* (Stoddard 1931), *The Bobwhite Quail: It's Life and Management* (Rosene 1969), *Population Ecology of the Bobwhite* (Roseberry and Klimstra 1984), and *Bobwhites in the Rio Grande Plain of Texas* (Lehmann 1984). Thus, it is unlikely that a lack of scientific information limits wildlife managers from restoring the species.

Instead, for the first several decades of the existence of the wildlife management profession, the bobwhite conservation community has operated in a manner that can be characterized as "anarchy." As a resident wildlife species, bobwhites are the

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sole jurisdiction and responsibility of as many as 39 sovereign, independent states. No overarching authority exists to provide unified vision, leadership, coordination, funding, technical services, and other fundamental functions. This institutional complexity contributed to a long period of inertia and stagnation coinciding with a corresponding lack of urgency, a high degree of disorganization, and myriad repetitively failing attempts to solve the problem with simplistic tactics.

Some of the tactics employed over the years were patterned after those that had proven successful for white-tailed deer, wild turkey and other restored species. In retrospect, it has become apparent that bobwhite restoration is a different and much more difficult challenge. For white-tailed deer and turkey, the solution largely involved trapping and relocating wild individuals from where they were to areas of vacant suitable habitat, then protecting the transplants until the new populations reached sustainable levels. In contrast, the central challenge confronting bobwhite restoration is landscape-scale habitat degradation. Wildlife managers have a weak track record of restoring widespread species diminished by habitat degradation.

A brief glimpse into the past illuminates the magnitude of the landscape-scale habitat challenge. Aldo Leopold conducted a statewide game survey of Mississippi in 1929, on contract with the Game Restoration Committee of the Sporting Arms & Ammunition Manufacturers' Institute. His unpublished February 1, 1929 "Report on a Game Survey of Mississippi" estimates that 85% of the land area of the state was suitable quail habitat. Bobwhite conservationists estimate in 2006 that less than 5% of the state now is suitable habitat. Leopold also reported official estimates that about 50% of the forested area of the state burned each year. He further cited unofficial estimates by local observers that 50 to 90% of uncultivated land burned every year. In contrast, only a token acreage of the state is burned annually today. These findings and contrasts are roughly applicable across much, if not most, of the core bobwhite range.

This extreme and widespread reduction in suitable habitat is evident from the U.S. Department of Agriculture (USDA) Natural Resources Conservation Service's (NRCS) National Resources Inventory. Across 22 core bobwhite states in 1997, there were some 210 million acres (85 million ha) of annually cultivated cropland, 120 million acres (49 million ha) of tame pasture and 35-40 million acres (14-16 million ha) of loblolly pine plantation. Each of these land-use types is so intensively converted and managed that the effective result is 370 million acres (150 million ha) of unsuitable habitat. This expansive conversion does not include additional tens of millions of acres (ha) of degraded (by overgrazing or brush encroachment, or both) rangeland; millions of acres (ha) of unmanaged closed-canopy hardwood and mixed pine/hardwood forestland; and the virtual elimination of fire as an ecologically beneficial force. The remaining areas of suitable bobwhite habitats across most of the historic range are so fragmented that many quail populations are isolated.

The well-known consequence of this intensive, widespread and long-term conversion and degradation of habitat is a significant (about 3.8% per year average, according to the Breeding Bird Survey, 1966 - 2005), widespread and long-term decline in bobwhite populations across most of the species' range. Populations of wild bobwhites apparently already have been extirpated from several states across the northern fringe of the historic range.

By the late 1990s, bobwhite conservationists were confronted with 2 basic choices. First, we could keep doing the same thing–anarchy by default–and accept the inevitability of widespread recreational extinction of bobwhites. Alternatively, we could do something radically different. In taking a bold different approach we obviously risk failure, but gain the only viable opportunity for meaningful success.

The NBCI Era

The psychological stimulus toward a radically different approach for stabilizing and restoring bobwhite populations arguably began with the publication of the North American Waterfowl Management Plan (NAWMP) in 1986. This continental restoration strategy was developed during the grim period of historically low duck populations throughout the 1980s. At the time, the NAWMP's ambitious population recovery goals, the calculated habitat restoration objectives and the $1.5 billion cost estimate seemed implausible. A decade later, major money for implementation was materializing from myriad unanticipated sources, habitat restoration was occurring by the hundreds of thousands of acres per year, and duck populations already were responding and recovering.

The first step in the transition from bobwhite anarchy to strategy may have been the Brennan (1991) paper in the Wildlife Society Bulletin, entitled, "How can we reverse the northern bobwhite population decline?", which raised widespread awareness with the prediction that bobwhites soon would become unhuntable across most of the species' historic range. Following that wake-up call, Tom Dailey and Kevin Church arranged a strategic quail planning workshop at the Quail III National Quail Symposium in Kansas City, Missouri in July 1992. The resulting "Strategic plan for quail management and research in the United States: introduction and background", edited by Brennan and published in the 1993 symposium proceedings, constituted the first attempt to develop a national framework for addressing declining populations for all six North American quail species.

The transition accelerated in August 1995 with the formation of the Southeast Quail Study Group (SEQSG). The charter meeting-initiated and organized by Breck Carmichael [South Carolina Department of Natural Resources (SCDNR)], hosted by the SCDNR, and sponsored by Quail Unlimited-was attended by 62 bobwhite conservationists from across the Southeast. The impetus for convening to form the SEQSG seems to have been a critical mass of coinciding awareness and interest among the directors and wildlife chiefs of the Southeastern Association of Fish and Wildlife Agencies (SEAFWA) and the participants in the 1993 strategic planning workshop. By the end of the 4-day inaugural meeting, the SEQSG was organized, a Steering Committee was appointed (with Carmichael as Chair), a series of working committees was formed and committee chairs selected. The SEAFWA Directors promptly endorsed the SEQSG and recognized it as a technical committee of the Southeastern Association.

The SEQSG quickly became the first unifying regional force to provide effective leadership for bobwhite conservation. Still lacking, however, was vision, purpose and direction.

The pivotal milestone that culminated the beginning of the strategic era for bobwhites was the autumn 1998 business meeting of the SEAFWA Directors. The directors, lead by Tennessee Wildlife Resources Agency Director Gary Myers, issued a verbal charge to the SEQSG to develop a regional, quantitative, habitat-oriented plan to restore bobwhites. A year later, the SEAFWA Directors clarified their charge by endorsing a goal of restoring bobwhites to 1980 levels. The long-term vision that was requisite for the SEAFWA Directors to make this watershed charge to the SEQSG could be the dawning of vital regional leadership on conservation of bobwhites and other resident wildlife species.

Following the charge from the SEAFWA Directors, the SEQSG Steering Committee tapped Ralph Dimmick, recently retired from the University of Tennessee, to lead the strategic planning effort. Dimmick assembled a core planning team of about a dozen professionals, but more than 50 biologists ultimately participated in development of the plan.

Although the SEAFWA represents 16 southeastern states, and the SEQSG is a technical arm of the SEAFWA, the members of Dimmick's planning team recognized the shortcoming of developing a bobwhite conservation plan that stopped at the artificial SEAFWA administrative boundaries. The planning team deliberately–but not lightly–decided to exceed its authority by including several non-SEAFWA midwestern "core" quail states for which basic data was available and in which at least one professional biologist was willing to participate. The plan ultimately covered 22 states.

A key paradigm-shifting early decision of the planning team was the recognition of the need to rise above the traditionally constraining view of the world within sideboards defined by politically based state boundaries. Instead, the planning team adopted the North American Bird Conservation Initiative's (NABCI) ecologically based "bird conservation regions" (BCRs) as the basic planning unit for bobwhite restoration. The NABCI, launched in 1999, was striving for unity, coordination and collaboration among all bird conservation initiatives by winning broad acceptance of common, ecologically based planning units. The planning team recognized the long-term value that adopting BCRs would secure for bobwhite conservation, itself, as well as the credibility, integration and partnerships that would be fostered with other bird initiatives.

The "Northern Bobwhite Conservation Initiative" (NBCI) was published March 2002, some 3 1/2 years after the initial charge from the SEAFWA Directors. The Initiative was promptly endorsed and adopted by the SEAFWA in May of that year, and by the International Association of Fish and Wildlife Agencies (IAFWA) in September 2002.

The NBCI vision to stabilize and then to restore bobwhites to 1980 levels was acknowledged to be a long-term task, requiring at least a quarter-century of sustained effort. This vision was specified by a restoration goal of adding about 2.8 million new coveys to the landscape above the 1999 populations. Achieving this bobwhite population increase would require restoration of suitable habitats on some 81 million acres (33 million ha) across 22 states. The NBCI framed BCR habitat objectives based on three major land-use types for which the technology exists to restore usable habitat for bobwhites: cropland, grazing lands and forest land (especially pine forests).

An important tenet of the NBCI is the determination that, across most of bobwhite range, the major limiting factor is inadequate recruitment, due to insufficient nesting and brood-rearing habitat. The NBCI defined this limiting habitat type as diverse stands of properly managed native, warm-season grasses with abundant forbs, legumes and wildflowers. Put another way, the NBCI calls for restoration of millions of acres of native grassland habitats in cropland, grazing land and forested landscapes.

A final key tenet is the overriding NBCI philosophy of the necessity to reconnect bobwhites with working agricultural lands. Because some nine-tenths of the land in the core bobwhite range is privately owned, bobwhite restoration ultimately is dependent on private lands for widespread success. Setting land aside and prohibiting human uses is neither necessary nor effective for bobwhites.

The first significant accomplishment of the new NBCI was the unprecedented unity of vision, mission, purpose and hope that it instilled quickly across most of the bobwhite conservation community. For the first time, most of the bobwhite community across numerous states was facing the same direction, agreeing on the need to pull together. This unity soon was bolstered with heightened interest and excitement. For example, the registered attendance at the annual August meeting of the SEQSG increased from 105 people in 2000 (the 2001 attendance is un-interpretable due to being postponed and combined with the Quail V conference) to an average of 185 in the years 2005 and 2006.

This feeling of community and excitement, in turn, translated into invigorated leadership and activity among the SEQSG and the SEAFWA. A coordinator position soon was proposed by SEAFWA Directors to foster implementation. The new NBCI Coordinator position was funded with a three-year Multi-state Conservation Grant from the IAFWA, supplemented with annual $2,000 contributions from each of the SEAFWA states plus $15,000 per year from the NRCS Wildlife Habitat Management Institute. Breck Carmichael was hired in January 2003, on loan from the SCDNR, to be the first NBCI Coordinator. When Carmichael was promoted to Deputy Director of SCDNR in summer 2004, the author was hired to fill the vacancy beginning in September 2004.

So many significant NBCI accomplishments have occurred in the first 4 1/2 years that it would be overkill to describe them all. The single biggest accomplishment has been the "CP33 Habitat Buffers for Upland Birds" practice created for the Continuous Conservation Reserve Program (CRP) in August 2004. The practice originally was conceived, written and proposed by the SEQSG in 1996, but soon was rejected by the Farm Service Agency (FSA). Following publication of the NBCI in 2002, the NBCI Coordinator, the author and the SEQSG refined the proposal and resubmitted it in July 2003. This time–with the backing of the NBCI and supportive managers' report language from the 2002 Farm Bill–not only did FSA accept the proposal, but President George W. Bush, himself, announced the new practice at a ceremony on a farm in Minnesota. FSA earmarked 250,000 acres (101,250 ha) of CRP authority to the CP33 practice across 35 states, a quail habitat practice worth more than $200,000,000. By the time of this conference, more than 115,000 acres (46,575 ha) of native-vegetation cropland field borders had been enrolled in 25 states.

Where Are We Now: Notable NBCI Accomplishments Since 2002

1. Bobwhite restoration now is a national issue.
2. Bobwhite restoration now is a priority for many states.
3. Bobwhite restoration and NBCI implementation were explicitly endorsed and supported by Congress in the 2002 Farm Bill Managers' Report.
4. Three $0.5 million federal appropriations were earmarked for bobwhite research, in fiscal years 2003 through 2005.
5. About 15 states now have active quail restoration initiatives.
6. NBCI stimulated creation of a position to represent resident game birds in the North American Bird Conservation Initiative.
7. NBCI stimulated IAFWA to create the Resident Game Bird Working Group.
8. Working relations with other conservation groups, such as Partners in Flight and the southeastern forestry community, are improving and purposeful.
9. The first-ever cooperative interstate bobwhite monitoring protocol was developed, funded and implemented in more than a dozen states for CP33.
10. A new longleaf pine practice has been developed and proposed for the Continuous CRP, and apparently is nearing approval.
11. NBCI is becoming an active presence in joint ventures.
12. The NBCI and CP33 were featured at the August 2005 White House Conference on Cooperative Conservation.
13. SEAFWA Directors created in May 2005 a "Directors' NBCI Committee", Chaired by Dan Forster (Director, Georgia Wildlife Resources Division).
14. A second Multi-state Conservation Grant was awarded in autumn 2005 from IAFWA to NBCI, to build interstate capacity:
    - hire a "National Advocate"
    - conduct regional and state NBCI step-down workshops
    - revise, upgrade, and expand the NBCI
15. Other bird initiatives are following the NBCI's lead approach:
    - woodcock
    - prairie grouse
    - ruffed grouse
    - western quail
16. USDA NRCS Plant Materials Centers are partnering with NBCI conservationists in 6 southeastern states to promote native, warm-season grasses.

17. Two non-government quail organizations are increasingly active, engaged and contributing.

18. Texas amended its state mineland reclamation regulations to promote quail habitat.

Much of the most important progress in NBCI implementation has, of necessity, occurred in the form of upgraded bureaucracy and infrastructure within state wildlife agencies. Even though the NBCI utilized BCRs as strategic planning units, the legal authority, responsibility and management capacity for implementing the NBCI still resides with the states. To aid the states with this task, the SEQSG developed a detailed questionnaire as a tool for internal use by state wildlife agencies, to provide specific guidance to states striving to better implement the NBCI, as well as to provide a mechanism for self assessment. The summarized results of an informal survey conducted of all 22 NBCI states by the author in 2006, based on eight questions selected from the assessment tool, are presented below.

Results Of Questionnaire To State Agencies On Implementation Of NBCI.

The state of implementation and administration of the Northern Bobwhite Conservation Initiative

1. *Has your state agency directorate and/or governing Board or Commission publicly committed that bobwhite restoration is a state priority?*

   10 of 22 states answered ”yes” (compared with ≈ 2 pre-NBCI).

2. *Does your state have some form of state-level inter-agency council, committee or task force focused on implementing the NBCI and restoring early successional/grassland habitats?*

   9 of 22 states answered ”yes” (compared with ≈ 0 pre-NBCI).

3. *Does your agency employ or have the benefit of a dedicated, full-time statewide quail program coordinator that is not distracted or diluted by other somewhat-related duties (e.g., small game coordination or farm bill liaison) but is focused on NBCI implementation and bobwhite restoration in the state?*

   5 of 22 states answered ”yes” (compared with ≈ 2 pre-NBCI).

4. *Does your agency employ or have the benefit of a dedicated, full-time statewide agricultural liaison or farm bill coordinator whose sole responsibility is to engage and improve state-level farm bill conservation programs for wildlife?*

   8 of 22 states answered ”yes” (compared with ≈ 6 pre-NBCI).

5. *Does your agency employ or have the benefit of an adequate and effective network of private lands biologists (PLBs) capable of, dedicated to and actually focused on delivering habitat conservation on private lands at a meaningful scale?*

   6 of 22 states responded that they have a very effective network of PLBs; 12 of 22 states responded that they have a somewhat effective network of PLBs (these 18 total PLB states compare with a total of ≈ 12 pre-NBCI).

6. *Has your agency officially designated and initiated concentrated quail habitat management efforts in one or more actual on-the-ground quail habitat restoration focal areas, especially including private lands?*

   14 of 22 states responded ”yes” (compared with ≈ 4 pre-NBCI).

7. *Has your agency ”stepped down” or tailored the NBCI to a more-detailed state-level bobwhite habitat restoration plan, with specific habitat objectives?*

   7 of 22 states answered ”yes” (compared with ≈ 0 pre-NBCI).

8. *Is your agency actively representing NBCI and/or state step-down bobwhite habitat objectives with the joint ventures and bird conservation regions in your state?*

6 of 22 states answered ”yes” (compared with ≈ 0 pre-NBCI).

Where Are We Going: Taking The NBCI To The Next Level

The SEQSG has informally adopted the theme “raising our game” to characterize and inspire collective efforts to build capacity and momentum necessary over the long term for the NBCI to succeed. The entire community of resident game bird conservationists has a limited track record of experience in collaborating across state and regional boundaries, when compared with the community of migratory bird conservationists. This lack of experience is a serious impediment when facing the need for effective interaction at multiple levels of society and government, including the federal government, regional associations of states across the bobwhite range, joint ventures, state governments, state-level coalitions, local governments and private landowners.

To further complicate the ability of bobwhite conservationists to raise our game, there is little existing infrastructure at any level to provide the capacity capable of dealing with these multiple layers of society and government. What infrastructure does exist–for example, the SEQSG–is not a good fit with the administrative challenges. For example, the SEQSG officially represents the 16 SEAFWA states, leaving the 6 midwestern NBCI states without formal representation. This administrative inadequacy inadvertently causes some states and bobwhite conservationists outside the administrative boundaries of the SEAFWA to consider the NBCI to be a southeastern initiative that is not relevant elsewhere.

Some meaningful steps are being taken to begin addressing a series of such serious administrative and infrastructure limitations. The first step was the extension of the NBCI Coordinator position for another three years, through spring 2009. The 16 SEAFWA states contributed $100,000 per year for three years, while Quail Unlimited pledged another $75,000 per year for three years.

The SEAFWA Directors’ NBCI Committee was expanded by two seats in 2006 to add official representation by a Director from both the Midwest Association of Fish and Wildlife Agencies (MAFWA) and the Northeast Association of Fish and Wildlife Agencies (NEAFWA). This measure already has borne fruit. The Directors of the Midwestern Association of Fish and Wildlife Agencies unanimously passed a resolution in 2006 endorsing the NBCI and pledging to support its revision and implementation. Likewise, the SEQSG Steering Committee was expanded by two seats to add official representation by technical staff from each of the MAFWA and NEAFWA.

The next major step is for the SEQSG to revise the NBCI, to ensure the plan stays current and continues to stimulate progress. Tall Timbers Research Station’s Bill Palmer has been contracted by the SEQSG to coordinate the national NBCI revision process. The NBCI will be revised based on the knowledge, experiences and attention gained in the first 4 years of implementation. For example, it will include additional states, address additional land-use types (such as western rangelands and mineland reclamation), include more professional participants, and feature improved assumptions and landscape analyses. The NBCI revision process officially began at the Gamebird 2006 conference in June 2006, with an open input session attended by about 55 people the first evening. The NBCI revision is expected to be completed by the end of 2007 or early 2008.

Conclusions

In its first 4 1/2 years, the NBCI has made more progress and achievements than anyone could have anticipated. Yet, the bobwhite community still is very early in what will be a long and challenging restoration process. Bobwhite restoration may be the most difficult challenge ever confronted by the wildlife management profession in North America; however, it is not a challenge of science. We have the technical knowledge on *how* to restore bobwhite habitats and populations. What we lack is the capacity to get it done on a large enough scale. Thus, bobwhite restoration is a challenge of will–*will* conservationists and society muster the resolve necessary to restore bobwhites?

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Restoring a Gray Partridge (*Perdix perdix*) Population and the Future of Predation Control

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For the past quarter century, the author's partridge (*Perdix perdix*) population simulation model has been a useful guide, incorporating annual variation in chick survival rate, nesting habitat quality, nest predation, pesticides affecting the supply of insect food for the chicks and shooting. The 2 most important variables, density dependent nest predation and effect of insect food supply have been experimentally verified and all parts of the model have been extensively validated. However some very high densities achieved in northern France have not been fully explained. An ongoing conservation research project within the Sussex Downs Study (U.K.) area is described. The area is small (155 ha) but the numbers of partridges have increased ten-fold since 2003, with 57 per 100-ha in the autumn of 2005. Despite a very high density of pheasants no adverse effect of a caecal nematode (*Heterakis gallinarum*) has been detected. This project has been successful so far, but it has involved the removal of large numbers of meso-predators. Although legal, this predation control is unpopular and difficult. Moreover the smaller raptors are increasing and cannot be controlled. The question of whether the return of some top-predators could improve the situation for the partridge is explored by reference to experience in the USA and other countries. Few species have been studied more than the partridge yet, even more vigorous research will be necessary in future to meet the many challenges ahead.

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Key words: gray partridge, Perdix perdix, predation, predation control, UK

For the past quarter century a deterministic simulation model of gray partridge (*Perdix perdix*) population dynamics has been a useful guide to the management of gray partridge populations in the UK, incorporating the dynamic interplay between density, annual variation in chick survival rate, nesting habitat quality, nest predation, insect availability for chicks due to pesticide use, and shooting (Potts 1986). The 2 most important variables, density dependent nest predation and effect of insect food supply have been experimentally verified and all parts of the model have been thoroughly validated. Adapted to local conditions, the model has also been found useful in North Dakota (Carroll 1992) and elsewhere (e.g. Germany Pethig 1994), but in France different factors have clearly emerged. In the Beauce (SW Paris Basin) predation on adult partridges by the hen harrier (*Circus cyaneus*) has become increasingly important (Bro et al. 2006). Yet, paradoxically in May 2000, at a meeting of the organization La Chasse Verte, Gérard Pasquet reported all time record densities of 1 pair/ha in the NW Paris Basin, particularly on 2 farms owned by Jacques Hicter. For these farms the Potts model predicted pair densities and over-winter survival that were half the observed levels, whereas the remaining variables were well predicted. Although analyses are not yet concluded it is clear that the almost year-round provision of grain on these two farms is reducing dispersal rates and thus elevating local densities. Due to these drastically different findings, there was considered to be a clear need to test these conclusions in the United Kingdom.

Partly with this in mind, in 2003/2004 a major new wild partridge conservation management and research project was established privately within the Sussex Downs Study Area (see Potts 1986). The part of the area chosen for a feasibility study was initially

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small (145 ha of arable), and intensive management involved the implementation of beetle banks, unsprayed [CH2] headlands, strips of kale (*Brassica oleracea*), nest predator control, supplemental feeding, a zero rate of gray partridge shooting and no rearing and releasing of gamebirds. The rest of the study area which totals 6200 ha remained as a control. However in spring 2006 the feasibility study was extended by 120 ha of arable with a further extension of 365 ha in spring 2007 to bring the intensively managed area to 630 ha of arable. During 2003-2006, this management package resulted in an increase to 15 pairs per 100-ha compared to 1-2 pairs per 100-ha in the remaining part of the study area (acting as a control) and in the managed area pre-treatment. In the managed area the large increase has taken place despite a very high density of pheasants (*Phasianus colchicus*) mostly originating from neighboring properties and with red-legged partridges (*Alectoris rufa*) also present. The project is privately funded and detailed results will be made available as equilibrium levels are established.

This ongoing partridge management project has like the previous Salisbury Plain Project (Tapper et al. 1996) required the removal of large numbers of predators. Although legal in the UK and France, this predation control is expensive, and necessitates a large amount of physical work in often difficult conditions. Moreover several species, especially the smaller raptors are increasing and these cannot be controlled. The question whether the return of some top-predators such as the eagle owl (*Bubo bubo*) could improve the situation for the partridge has been explored (Potts 2007). For example in Schleswig Holstein in the absence of eagle owls it was found in one study that the northern goshawk (*Accipiter gentilis*) predated about 13% of adult female gray partridge per year. The eagle owl also takes many partridges; it is increasing virtually throughout Europe and thus may add to the predation pressure on the gray partridge. However calculations suggest that the additional predation on partridges by these owls is more than offset by its predation on partridge predators. Many questions remains unanswered but intra-guild predation like this may help to relieve the intense nest predation pressure that most partridge populations currently experience on farmland and more research here would clearly be valuable.

Few species have been studied more than the partridge, yet the continually evolving dynamic and complex interaction of gray partridges with their ecosystem demonstrate that even more vigorous research will be necessary in future to ensure the species has a satisfactory future.

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Abundance Estimation

The GWCT Partridge Count Scheme: a Volunteer-Based Monitoring and Conservation Promotion Scheme

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The Game & Wildlife Conservation Trust's (GWCT) Partridge Count Scheme (PCS) is a volunteer-based monitoring system serving as a means for delivering conservation and 'best practice' advice to participants (farmers, land managers and gamekeepers). Originally designed to monitor numbers of grey partridge (*Perdix perdix*) on UK shooting estates, it has been expanded to include participants primarily interested in conservation in response to the UK government's Grey Partridge Species Action Plan. The PCS is also an invaluable tool for examining trends in partridge abundance and population parameters. We examined trends in annual Grey Partridge production (autumn counts available from 1933) and breeding abundance (spring counts available from 1952). We compared trends of production and breeding abundance from old participants to trends from recently joined participants and interpreted the results relative to the Grey Partridge Species Action Plan. We also discuss the provision of data back to PCS participants and future plans for conservation advice and collaboration with other organizations, in particular the UK government's Department for the Environment, Farming and Rural Affairs.

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Key words: gray partridge, Partridge Count Scheme, *Perdix perdix*, UK

Introduction

The Game & Wildlife Conservation Trust's Partridge Count Scheme (PCS) collates information on the annual abundance and breeding success of grey partridges based on counts of pairs in spring and counts of young and old birds in the autumn by volunteer contributors to the scheme. From 1933 to 1998, the scheme mainly involved around 100 shooting estates in eastern and southern England (Potts 1980, Potts and Aebischer 1995). It had evolved from work carried out across a number of grey partridge shooting estates in the 1930s by staff at the Bureau of Animal Population at Oxford University (Middleton 1934, 1935*a*). Data was not only collected on abundance and breeding success - measured from bird counts - but on many of the estates the fate of individual nests was monitored (Middleton 1935*a*). As pheasants (*Phasianus colchicus*) have replaced grey partridge as the main quarry species in lowland Britain (Tapper 1992), nest finding as a gamekeeper activity is now a rarity. Calculations of grey partridge population parameters are now used to give a measure of nesting success (Potts 1986, Potts and Aebischer 1995), based on work begun using information from the early PCS (Middleton 1935*b*) and other sources (Blank and Ash 1958, Blank et al. 1967).

From early on information gained through the PCS was used to draw conclusions on grey partridge numbers across the UK's shooting estates (Middleton 1934, 1935*b*, 1937). Using this data, publications of the time highlight the same concerns expressed more recently when discussing declines in grey partridge numbers (loss of gamekeepering - Middleton 1947, Potts 1980), (the effects of intensive farming - Middleton and Ash 1964, Potts 1980, Potts and Aebischer 1995).

Recent expansion of the PCS arose in response to a renewed interest in addressing the continued decline of grey partridge numbers in the UK (Marchant et al. 1990). With being named 'lead partner' for the Grey Partridge Biodiversity Action Plan (Anony-

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mous 1995) the Game & Wildlife Conservation Trust (GWCT) launched the Partridge Recovery Program, with three main strands: raising awareness, setting targets and motivating by example (Aebischer 2009). As part of this, The GWCT made the decision in 1998 to expand the membership of The PCS. The expanded PCS would provide practical support and advice to farmers and landowners, who would need to undertake the management necessary to meet the targets and also give some means of monitoring progress towards the BAP targets on land within the scheme. We report the results of that expansion, recent trends in count data and illustrate how we are utilizing the PCS as one of the strands in the Partridge Recovery Program.

Study Area

The Partridge Count Scheme (PCS) membership is comprised of farmers, landowners and shooting syndicate members throughout Great Britain, with the intention of undertaking partridge counts on land under their management. Prior to 1999, most members had an interest in the shooting of grey partridges, either actively or in the recent past. In 1998 the PCS was expanded to Great Britain as a whole and currently (spring 2006) there are 1889 registered participants within the scheme.

Methods

Target Setting

Maps showing the extent of the area to be counted are requested when a contributor registers with the PCS; these are digitized into a GIS (Mapinfo Version 8) and are used to provide an individualized BAP targets for each count area. Targets are calculated using the method outlined in Tapper (1999), revised by Aebischer (2009). Calculating targets based on landscape characteristics allowed us to set both local (farm) and county-level targets so that farmers and local government can assess progress in their area towards the UK-wide goal of 150,000 pairs in 2010.

Data Collection Protocol

The Partridge Count Scheme (PCS) database contains information from autumn stubble counts of grey partridge and red-legged partridges from 1933 to 2005 and from 1951 to 2006 for spring pair counts. All counts are carried out by volunteers - usually the gamekeepers, farmers, managers or owners of the shoots, farms and estates who are registered with the PCS. All counters are encouraged to follow a standard method of counting as per Potts (1986) with a recommendation that counts are not undertaken in winds stronger than Beaufort Force 3 (Gentle Breeze - Leaves and small twigs in constant motion; wind extends light flag). Spring counts take place in March/April and the number of both pairs and single birds are noted. Autumn counts are undertaken post-harvest from late August through October, with the number of males, females and young in each covey recorded. The longitude and latitude of the centre of each counted area was also recorded (in British National Grid projection). Information from the counts is returned to The Game & Wildlife Conservation Trust's PCS coordinator and then entered into a Microsoft 2003 Access database.

Information on seven different types of habitat management undertaken on the contributing estates has been collated for areas where counts were undertaken in both spring and autumn of 2005 - the first year where this data is available. This includes whether or not an estate had any of the following: conservation headlands, beetle banks, uncut grass margins, additional food provided, typically grain, in either autumn or spring, game cover crops, planted as either brood-rearing or over-winter cover. We also recorded information on whether or not grey partridges had been released for either conservation or shooting. Additionally, the density of gamekeepers (per km^2) was calculated, as was the shooting pressure (proportion of the grey partridges available in autumn that were shot).

Statistical Analyses

We compared the farm/estate target and the number of spring pairs counted in 2005. For those estates who had a target of zero - no suitable habitat for grey partridges - we only considered that they had reached their target if they had counted at least one pair of partridges in the spring of 2005, otherwise they were discounted from the analysis.

Chi-square analysis was used to compare the number of new and long-term contributors that released partridges, had implemented the seven surveyed habitat managements, and the proportion of each that had successfully met their BAP targets. A t-test of the ln-transformed area counted (transformed to stabilize the variance) and the proportion of the autumn stock shot (transformed to angles) was used to compare the size of the counted areas and the relative effect of shooting on the two types of contributors.

A generalized linear model with Poisson error logarithmic link function and ln(area counted) as offset to the number of gamekeepers was used to compare the density of gamekeepers and of the number of birds shot over the area counted between long-term and new contributors. As count data are not normally distributed, we used a generalized linear model with Poisson error logarithmic link function and ln(area counted) as offset of the number of spring pairs recorded in the spring of 2005 to determine which of the seven surveyed types of habitat management as well as shooting pressure, had a significant effect on abundance (Aitkin et al. 1992, Dobson 2002, Seavy et al. 2005). We controlled for whether or not a site was a long-term or new contributor, had released partridges, geographical location (entered as the interaction of easting and northing) and gamekeeper density. Forward stepwise selection (at $P < 0.05$) was used to select the most parsimonious model. A similar approach, using a generalized linear model with binomial error was used to determine which, if any, of the seven surveyed managements influenced whether or not a site met its BAP targets in 2005, again controlling for type of contributor, geographical location and gamekeeper density. The generalized linear modeling was carried out in Genstat version 8.2 (Lawes Agricultural Trust), with Systat version 10 (SPSS Inc.) used for t-tests and chi-square analysis.

Analysis of Trends in Abundance and Production

Annual indices of abundance were calculated by fitting a generalized linear model with Poisson error logarithmic link function and ln(area counted) as offset to the number of spring pairs recorded from 1951 to 2006, using site and year as factors, including only those sites that had returned counts for more than one year. Separate models were fitted to long-term and new contributors and these were compared for those years (from 1999) where data was available for both, using likelihood ratio tests, adjusted for over dispersion. The average young-to-old ratios indices were calculated as annual weighted means using the number of old birds as weights and analyzed by weighted analysis of variance. Previously (Potts 1980, Potts and Aebischer 1995) annual chick survival rates derived from autumn counts has been used to measure annual production, but the steady decline in the number of broods due to declining numbers necessitated a switch to young-to-old ratio as a measure of production.

From 1999 onwards, trends in the calculated annual indices for abundance and young-to-old ratios were examined for long-term and new contributors separately, using linear regression, weighted by the number of counts in each. For long-term contributors, a linear trend was compared to a quadratic trend for densities from 1995 (the beginning of the BAP). All generalized linear modeling and analysis of variance was carried out in Genstat version 8.2 (Lawes Agricultural Trust), with Systat version 10 (SPSS Inc.) used for the linear regression of annual indices.

Results

The expansion in the PCS membership is obvious when the number of returned counts in both spring and autumn is examined on a yearly basis (Figure 1). Although limited autumn counts were available before the 2nd World War, significant participation occurred from 1957 with spring counts in-

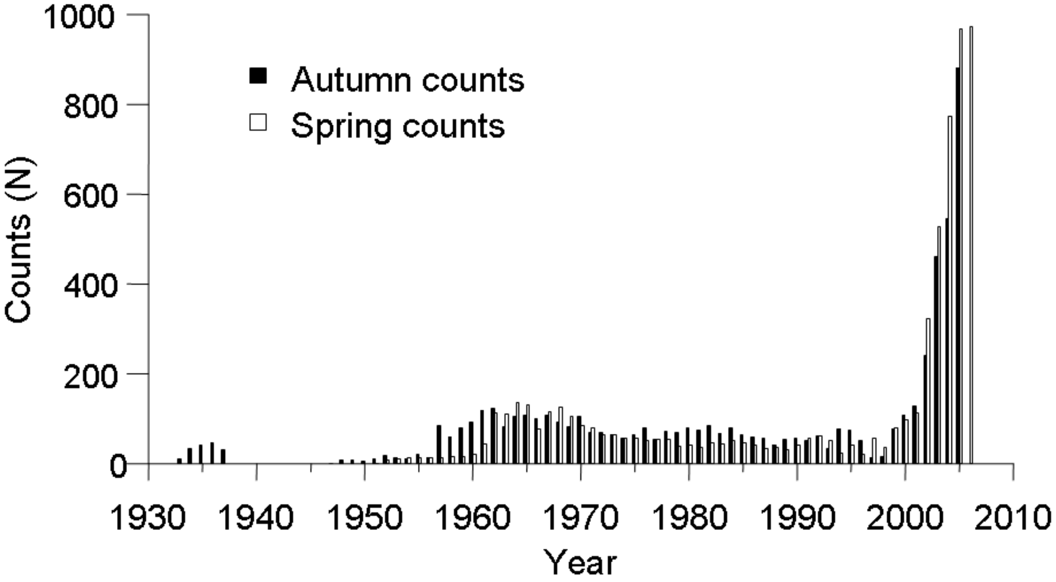

Figure 1: Increases in the number of both spring and autumn counts returned for members of the Partridge Count Scheme.

creasing in 1962. The number of submitted counts was relatively high in the 1960s, but only an average of 40 counts/year were returned in the mid-1990s. The success of the recent expansion of the scheme is evident, with a steady increase in counts returned from 2002 onwards. In 2005, the last completed year, 967 spring counts and 882 autumn counts were returned. The total area covered by the spring counts in 2005 was 3,165 km^2 on mainly cropped land in lowland Britain; this works out at roughly 5% of the UK arable area (total of 57,770 km^2 - Department for Environment, Food and Rural Affairs 2005*a*). A total of 9,752 grey partridge spring pairs were counted, comprising 15% of the 65,000 spring pairs that is the estimated UK population (Aebischer 2009).

Comparing Contributors

We compared the size and habitat and partridge management that was undertaken by long-term and new contributors who returned counts in both spring and autumn 2005 (Table 1). Sites that were long-term contributors were larger, had higher densities of gamekeepers, and their managers were more likely to undertake spring feeding and grey partridge releases for the purposes of conserving grey partridges and to established un-mown grass banks and conservation headlands. This may support the assumption that the long-term contributors had been undertaking at least some of the management needed for grey partridges, prior to the expansion of the scheme in 1999. Relative shooting pressure is similar on the two types of contributors, though the actual number shot on the areas managed by the long-term contributors is higher.

Management, Abundance and BAP Targets

We found no difference between long-term and new contributors in the proportion that reached or exceeded their individualized BAP targets (χ^2 = 2.87, P = 0.090), with 58% overall meeting or exceeding their individualized targets. In 2005, higher spring pair densities were found on areas where spring feeding ($F_{1,568}$ = 9.34, P = 0.002) and the planting of brood-rearing cover took place ($F_{1,568}$ = 11.41, $P < 0.001$) and lower densities were found on areas that planted over-winter cover ($F_{1,568}$ = 5.17, P = 0.023), after controlling for geographical location ($F_{1,568}$ = 49.81, $P < 0.001$), partridge releasing

Table 1: Characteristics of the counted areas from long-term and new PCS contributors, with comparisons in the management undertaken on the two different types of contributors where both spring and autumn counts were undertaken in 2005.

		Contributer Type		Statistics	
		Long-standing	New		
N		75 (10.8%)	620 (89.2%)	$P < 0.001$	$t_{691} = 8.12$
Area (km^2)		5.9	2.9	$P = 0.905$	$F_{1,504} = 0.01$
Autumn Stocks Shot		12.5%	13.5%	$P = 0.106$	$t_{493} = 1.62$
Number Shot/km^2		2.4	1.3	$P < 0.001$	$F_{1,504} = 14.35$
Gamekeeper/km^2		0.19	0.15	$P < 0.001$	$\chi^2 = 86.68$
Releasing	Conservation	25 (33.8%)	37 (6.0%)	$P < 0.001$	$\chi^2 = 86.68$
	Shooting	3 (4.1%)	23 (3.7%)	$P = 0.411$	$\chi^2 = 0.68$
Over-winter	Cover	40 (53.3%)	402 (64.8%)	$P = 0.141$	$\chi^2 = 2.16$
	Autumn Feeding	55 (73.3%)	434 (70.0%)	$P = 0.252$	$\chi^2 = 1.31$
Nesting Period	Spring Feeding	56 (74.7%)	385 (62.1%)	$P = 0.010$	$\chi^2 = 6.59$
	Uncut Grass Margins	56 (74.7%)	412 (66.5%)	$P = 0.042$	$\chi^2 = 4.13$
	Beetle Banks	28 (37.3%)	164 (26.5%)	$P = 0.101$	$\chi^2 = 2.68$
Chick Rearing	Brood-rearing Cover	27 (36.0%)	222 (35.8%)	$P = 0.514$	$\chi^2 = 0.43$
	Conservation Headlands	45 (60.0%)	294 (47.4%)	$P = 0.023$	$\chi^2 = 5.16$

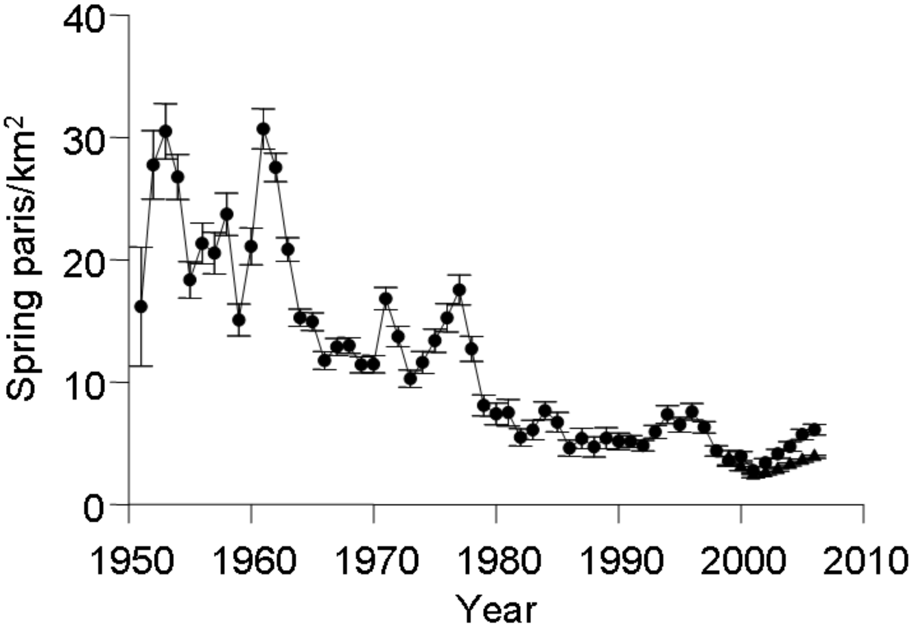

Figure 2: Trends in grey partridge abundance for long-term and new contributors to the PCS. The abundance reported by long-term contributors is higher than that of new contributors.

($F_{1,56}$ = 5.93, P = 0.003), gamekeeper density ($F_{1,568}$ = 0.01, P = 0.993) and whether or not an area was a long-term or new contributor to the PCS ($F_{1,571}$ = 51.31, $P < 0.001$). None of the seven surveyed habitat managements or shooting pressure were significantly associated with whether or not a contributor reached their BAP target, after controlling for geographical location ($F_{1,526}$ = 31.15, $P < 0.001$), releasing ($F_{1,526}$ = 1.51, P = 0.222), gamekeeper density ($F_{1,526}$ = 1.94, P = 0.164) and whether or not the area was a long-term or new contributor ($F_{1,526}$ = 2.51, P =0.113) to the PCS (Table 1).

Changes in Grey Partridge Abundance

The information on grey partridge numbers provided by the members of the PCS scheme is one means of monitoring grey partridge abundance. The annual changes in the abundance of grey partridge (counted spring pairs/km^2) from 1999 to 2006 were compared between old and new contributors (Figure 2). The pattern of change between the long-term and new contributors did not differ, ($F_{28,4491}$ = 0.63, P = 0.937), though long-term contributors had a higher abundance than the new contributors ($F_{1,2376}$ = 1631.71, $P < 0.001$). Annual indices were calculated for new and long-term contributors. New and long-term contributors showed differing trends in annual indices of abundance over the short time period (1999 to 2006) that data exists for both (comparison in trends - $F_{1,12}$ = 5.57, P = 0.036). Restricting the analysis to those years since increases in abundance began (2000 to 2006), showed no significant difference in the trends of long-term and new contributors ($F_{1,10}$ = 4.55, P = 0.059), with the abundance on long-term sites significantly higher than those on new sites ($F_{1,11}$ = 17.77, P = 0.001). The abundance reported by long-term contributors has increased by an average rate of 19% while that of the new contributors has increased by 8%. The annual abundance reported by long-term contributors since 1995 -beginning of BAP - did not fit a linear model ($F_{1,10}$ = 0.97, P = 0.347) but did fit a quadratic model, ($F_{2,9}$ = 18.38, P = 0.001), indicating that on these sites over this time period there has been a decrease and then

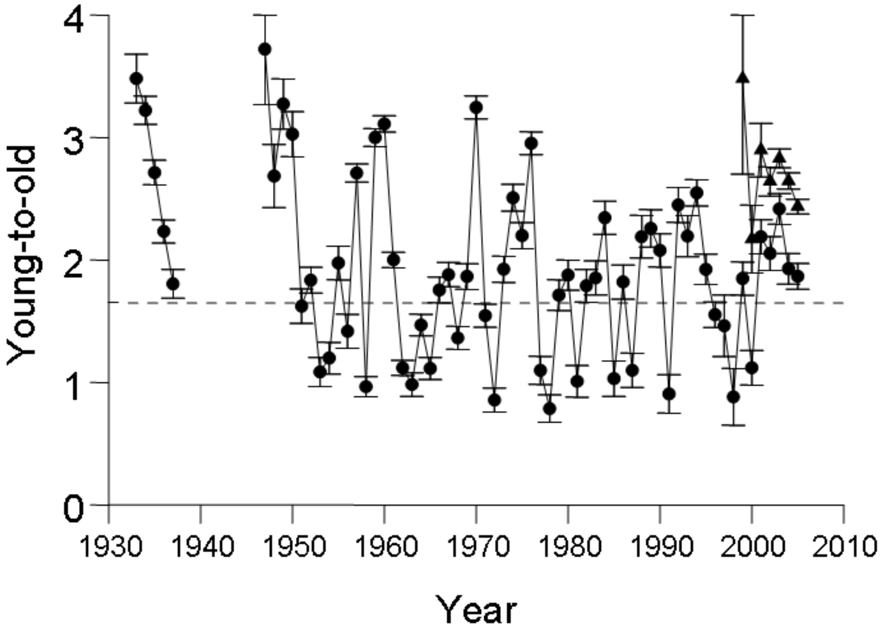

Figure 3: Trends in young-to-old ratio for long-term and new contributors to the PCS. The level needed (1.65 on average) to maintain grey partridge densities is shown as a broken line.

increase in abundance.

Trends in Grey Partridge Yearly Production

Grey partridge annual production, measured as young-to-old ratio, was compared between old and new contributors that had returned more than one count (Figure 3). The pattern of change did not differ significantly between new and long-term contributors ($F_{21,4502}$ = 0.43, P = 0.988) with the new contributors having higher young-to-old ratios than the long-term contributors ($F_{1,1449}$ = 56.58, $P < 0.001$). There were no trends in young-to-old ratio over the last six years for either type of contributor ($F_{1,11}$ = 0.01, P = 0.978), though the annual indices for new contributors were higher than those of old contributors ($F_{1,11}$ = 16.19, P = 0.002). From 2001, annual indices of young-to-old ratios for both types of contributors have been above the 1.65 level that is a prerequisite for stability in grey partridge numbers - dependent on levels of over-winter survival (Potts and Aebischer 1991).

Discussion

Expanding the PCS has increased the number of land managers who receive information about grey partridge research and advice on how to increase the number of grey partridges by 20 fold. This expansion contributes towards one of the key objectives set for Lead Partners of BAP species (raise awareness and promote management that will address declines in the BAP species of interest). This is an important consideration as changes will have to be made on farmland across the whole of the UK, not on a few scattered shooting estates to address the long-term widespread declines in grey partridge. Officials from England's Department for Environment, Food and Rural Affairs (DEFRA) are using some of the local partridge groups as "points of contact" to actively target farmers applying for agri-environment schemes, specifically Countryside Stewardship and the Higher Level Scheme within the new Environmental Stewardship (ES) options (Department for Environment, Food and Rural Affairs 2005*b*). Membership of the PCS is viewed in a

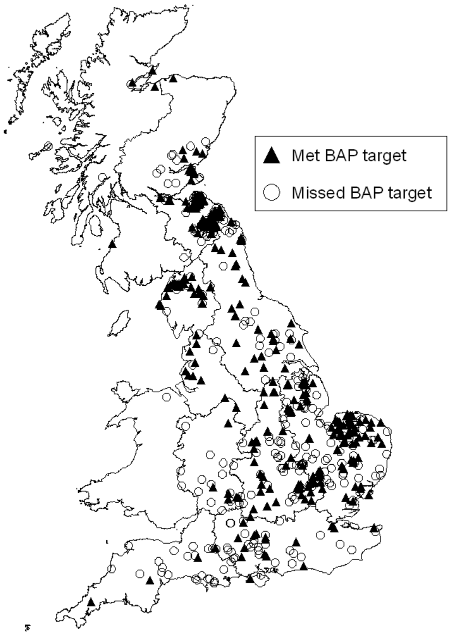

Figure 4: Distribution of counted areas in spring 2005, labeled as to whether or not they met their BAP targets.

positive light by DEFRA officials and members' applications for grants to cover management for grey partridges are given a higher priority. An example of this type of cooperation is found in Cumbria, a county in northwest England - where the local DEFRA official has encouraged the recipients of Countryside Stewardship grants to join the PCS. In 2005 (Figure 4) it is noticeable that a majority of PCS members in Cumbria met their BAP targets, something that was not necessarily the case in other western counties of England.

Had the members of the PCS met the initial target of halting the decline of grey partridges on their areas by 2005?

An examination of the trends in spring counts for both types of contributors from 2001 onwards indicate increases in abundance from that time (Figure 2). On the areas managed by PCS contributors, the decline has been halted and there are the beginnings of a recovery. How does this relate to Great Britain as a whole? A comparison of the percentage of Great the UK counted through the PCS in 2005 (5%) and the percentage of the expected current populations of grey partridges counted (15%) underlines the fact that the members of Partridge Count Scheme do not represent a random sample of farms and estates across the UK; they are a self-selected interested minority. As such they will have made more of an effort on halting the decline, so might be expected to be a best case scenario of what is happening across the UK as a whole. Comparisons with other national monitoring schemes may suggest that trends seen in the long-term contributors of the PCS are reflected in these other schemes, with some evidence of leveling off in grey partridge declines (Aebischer 2009, Raven and Noble 2006).

What do the results from the PCS monitoring suggest in regard to the second BAP target, that of reaching 150,000 pairs by 2010?

To answer this question we must extrapolate from the measured yearly increases in abundance from 2000 to 2006 on areas managed by PCS contributors. As new contributors differed in their rate of increase from the long-term contributors and also as they report undertaking less management, their rate of increase is a perhaps a maximum estimate of what is possible across the country as a whole. If the current national population of grey partridges is 65,000 pairs (Aebischer 2009), and the population grows at 10% a year (best case scenario from above), this results in over 90,000 pairs in 2010. The same rate of growth results in over 150,000 pairs in 2015. Clearly the answer to above question is no. Even if increases in the grey partridge national population were to begin in 2006, current evidence from the PCS suggests it is extremely unlikely that the 150,000 target will be met by 2010 and gives one estimate - 2015, admittedly a best case scenario, of when this target might be met.

What type of management is the most effective at meeting the BAP targets?

From our analysis of the management undertaken by PCS contributors, no single type of management stands out as being particularly related to whether or not an individual contributor's BAP target is met. It is hoped that the widespread availability of options under the new ES schemes, such as the use of Conservation Headlands (Sotherton et al. 1993), Beetle Banks (Thomas et al. 2001), the planting of brood-rearing cover under the Wild Bird Seed Option, could make management that will benefit grey partridge more commonplace throughout Britain. This supports previous modeling work that indicated that reaching the target of 150,000 pairs would require increasing both the amount of insect-rich habitat and nesting cover (Aebischer and Ewald 2004). This should be borne in mind by farmers undertaking management; it is important to select options that fulfill both requirements. Our results however, do not take into account the length of time these management options have been in place, as information on habitat management has only really begun for PCS contributors.

It is apparent that a high proportion of PCS contributors in the West Midlands and South Central England failed to meet their BAP targets (Figure 4), indicating that more effort needs to be applied in these areas. We have set up local partridge groups in these areas (Aebischer 2009) and it is hoped that advice on a local level may be effective in improving grey partridge densities here. Large parts of the very southwest of England, Wales as a whole and western and northern Scotland have few or no active PCS members. We need to recruit more PCS members in these areas at the fringes of the range of the grey partridge (Gibbons et al. 1993).

Management Implications

The main implication from this work is that even though the first UK BAP target for grey partridge has been met on areas managed by PCS members and may also have been met nationally (Aebischer 2009, Raven and Noble 2006), it is highly unlikely that the second target - 150,000 pairs in the UK by 2010 - will be. This will require sustained effort on the part of conservation bodies, farmers and agricultural policy makers. The expanded PCS contributes towards this and will continue to do so, demonstrating the utility of volunteer-based monitoring programs in the conservation of declining species.

Acknowledgments

We would like to thank all of the contributors to the Partridge Count Scheme over its long history, both for their count information and the work they do to conserve grey partridges. Staff at the Game & Wildlife Conservation Trust contribute to the smooth running of the PCS and provide support to the local partridge groups. In particular we would like to thank N.J. Aebischer, S. Browne, J. Daplyn, N. Deckker, L. Ferguson, N. Graham, I. Lindsay, M. McKendry, N. Sotherton, H. Straker, M. Swan, P. Thompson, and M. Tickler. N.J. Aebischer commented on a draft of this manuscript and offered statistical advice. G.R. Potts ensured the continuation of the PCS through the 1970s into the late 1990s and, together with E. Darling (of Green Globe Consultancy), was involved in the initial expansion of the PCS in the late 1990s.

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Refining the Morning Covey-Call Survey to Estimate Northern Bobwhite Abundance

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Morning covey-call surveys have been mentioned extensively as a practical and efficient way to estimate abundance of northern bobwhite (*Colinus virginianus*) with minimal effort. However, many of the protocol specifics such as radius of audibility and probability of a covey calling were not based on empirical data. Other limitations also existed such as variation in observers' detection capacity and calling behavior of coveys. These shortcomings limit the reliability of this survey technique. The objectives of our study were to 1) obtain an empirical estimate of radius of audibility, 2) document observer variability in estimating number of coveys heard, and 3) document calling behavior of bobwhite coveys. We found no difference in radius of audibility for areas with low-brush density (6%; 956 ± 72m; mean ± SE; *n* = 4 observers) and high-brush density (30%; 931 ± 66 m). We calculated an overall radius of audibility of 900 m (254 ha) pooled across sites. We documented considerable observer variability (CV 18-49%; *n* = 12 observers) in detecting coveys. Regarding calling behavior, we observed that 67% ± 9.0 (*n* = 30 coveys) of coveys emitted the covey-call during 2004 whereas 88% ± 7.8 (*n* = 17 coveys) of coveys emitted the covey-call during 2005. Of the coveys that called, 70% ± 10.2 in 2004 and 93% ± 6.4 in 2005 involved >1 bird calling/covey. Our findings allow for a refinement of the morning covey-call survey. We recommend obtaining site-specific radius of audibility and using a core number of observers that remain consistent from year to year.

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Key words: morning covey call, northern bobwhite, radius of audibility, survey

Introduction

Northern bobwhites (*Colinus virginianus*; hereafter, bobwhite) are highly social animals whose communication repertoire consists of a wide variety of calls (Stoddard 1931, Stokes 1967). Calls can be classified into general categories based on purpose: reproduction, group movement, and predator avoidance (Collias 1960). The most frequently heard calls regard reproduction and group-movement. Reproduction calls include the familiar *bob-white* and caterwauling calls (Stoddard 1931, Stokes 1967). These calls are issued primarily by males during the breeding season to attract mates or communicate with other males (Stoddard 1931, Stokes 1967). The most commonly heard group-movement call is the covey-call (a *koi-lee* or *hoy*) and is used for reuniting scattered coveys (Stokes 1967). Covey-calls can be heard during daylight hours after flushing a covey and during early-morning hours when coveys leave their night roost at the onset of sunrise (Stokes 1967). Guthery (1986) speculated that bobwhites issued the call at dawn to reunite coveys that had been scattered during the night. From a management perspective, the *bob-white* and *koi-lee* calls have been particularly helpful because they have allowed for estimation of bobwhite abundance (Stoddard 1931, Bennitt 1951, Guthery 1986, DeMaso et al. 1992, Seiler et al. 2002, Wellendorf et al. 2004).

The use of the covey-call as a potential method to index bobwhites was first proposed by Stoddard (1931). He proposed that counting morning covey calls could be used as an alternative method of estimating covey abundance when dogs were not available. The method was subsequently referred to by Roseberry (1982) as a way to index relative abundance but noted that the method was not sufficient

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to provide a measure of absolute abundance. The first reference identifying the use of morning covey calls as a method to estimate density appears to be Davis (1979). He noted that if the mean covey size was obtained, morning covey calls could be used to estimate absolute density by assuming a survey area of 200 ha. Since the Davis (1979) publication, several studies have attempted to validate or refine the morning covey-call survey as a way to estimate bobwhite abundance (DeMaso et al. 1992, Seiler et al. 2002, Wellendorf et al. 2004).

The general protocol of the morning covey-call survey involves establishing listening posts within the area to be surveyed and counting the number of coveys heard calling at these posts during dawn (Guthery 1986). Observers typically arrive at listening posts about 45 minutes prior to sunrise (DeMaso et al. 1992), and once coveys begin calling, observers mark the approximate location and distance of each call (Guthery 1986). Surveys are conducted for about 10-20 minutes beyond the timing of the last call (Guthery 1986, Seiler et al. 2002). Abundance is obtained by converting the index (coveys heard/point) to density (bobwhites/area) using an estimate of radius of audibility and mean covey size. Mean covey size can be readily obtained from field flushes of coveys; however, the radius of audibility is unknown. Estimates have ranged from 400 m (Roseberry 1982) to 15 km (DeMaso 1991).

Morning covey-call surveys have been used extensively to provide an estimate of bobwhite abundance (Roseberry 1982, Guthery 1986, DeMaso et al. 1992, Seiler et al. 2002, Wellendorf et al. 2004). This technique is practical and efficient for estimating densities on large areas with little effort. Guthery (1986) recommended 1 listening post per 800 ha and estimated minimal associated cost ($10-$15/listening post/day). However, because specifics of the technique have not been quantified (e.g., radius of audibility), application of the technique has been limited. Given the extensive and practical use of the technique, the objective of our study was to refine the underlying protocol of the covey-call survey. Specifically, our objectives were to 1) obtain an empirical estimate of the radius of audibility, 2) document observer variability in estimating number of coveys heard, and 3) document calling behavior (i.e., probability of calling and number of individuals/covey calling) of bobwhite coveys.

Study Area

Our study was conducted on 2 study areas in southern Texas: the Encino (Brooks County) and Santa Gertrudis (Kleberg County) Divisions of King Ranch, Inc. The Encino division was located within the Rio Grande Plains ecoregion (Gould 1975) of Texas. This ecoregion is characterized by level to rolling land that is dissected by streams flowing into the Rio Grande or the Gulf of Mexico (Scifres 1980). The average annual rainfall in this area is 40-76 cm with the greatest amount in May and June and the least in January and February (Correll and Johnston 1979). On the Encino division, we used 3 pastures (North Viboras, Loba, and Cuates) for our study. A woody cover gradient existed from north to south with the northernmost pasture, North Viboras, consisting of the greatest brush coverage ($\approx 30\%$) and the southernmost pasture, Cuates, having the least ($\approx 5\%$). Vegetation common to all 3 pastures consisted predominately of honey mesquite (*Prosopis glandulosa*), live oak (*Quercus virginiana*), granjeno (*Celtis pallida*), pricklypear cactus (*Optuntia lindheimeri*) and huisache (*Acacia smallii*). Common grasses consisted of little bluestem (*Schizachyrium scoparium*), paspalum (*Paspalum* spp.), Kleberg bluestem (*Dichanthium annulatum*), King Ranch bluestem (*Bothriochloa ischaemum*), buffelgrass (*Pennisetum ciliare*), red lovegrass (*Eragrostis secundiflora*), gulf cordgrass (*Spartina spartinum*), and sandbur (*Cencchrus incertus*). Predominant forbs included croton (*Croton* spp.), dayflower (*Commelina erecta*), partridge pea (*Chamaecrista fasciculate*), and sunflower (*Helianthus annuus*). Predominant soil types were Falfurrias, Sarita, Sauz, Loba, Quiteria, and Padrones fine sands (United States Department of Agriculture 1993).

The Santa Gertrudis division was located in the Gulf Coast Prairies and Marshes ecoregion (Gould

1975). The topography of this ecoregion is generally level with poorly drained soils (Scifres 1980). The average annual rainfall in this area is 76-127 cm with the greatest amount in May and June and the least in January and February (Correll and Johnston 1979). Brush species present in Santa Gertrudis Division were honey mesquite, live oak, lotebush (*Ziziphus obtusifolia*), blackbrush acacia (*Acacia rigidula*), Texas persimmon (*Diospyros texana*), brasil (*Condalia hookeri*), colima (*Zanthoxylum fagara*), granjeno, and huisache. Predominant grass species included silver bluestem (*Bothriochloa laguroides*), knotroot bristlegrass (*Setaria geniculata*), plains bristlegrass (*Setaria leucopila*), Kleberg bluestem, buffelgrass, and Texas cottontop (*Digitaria patens*).

Methods

Radius of Audibility

RECORDING OF COVEY-CALL - In order to conduct audibility trials, we first obtained digital recordings of bobwhite coveys under natural field conditions during March 2005. We used radio-marked bobwhites from an ongoing, long-term study (South Texas Quail Research Project) to monitor covey behavior and calling. For the recording, we randomly selected 3 radio-marked bobwhites from the sample (n = 60) and located bobwhites to within 20 m 45 minutes before sunrise. When the covey-call was emitted, we recorded the call using a Sennheiser microphone (Sennheiser Electronic Corporation, Old Lyme, Connecticut) connected to a digital audio tape (DAT) recorder (Sony Corporation of America, New York, New York) and power source (Rolls PB223 Dual Phantom Power Adapter, Rolls Corporation, Murray, Utah).

We also estimated loudness (decibels) of the covey-call from captive birds. We estimated loudness of the call using a digital sound level meter (Extech 407735, Extech Instruments, Waltham, Massachusetts). We obtained estimates of the loudness of the covey-call from captive bobwhites on 3 separate occasions in order to obtain sound readings nearest the source as possible. We recorded the maximum decibel reading observed and the distance to the calling bobwhite.

DELINEATION OF TRIAL PLOTS - We attempted to conduct audibility trials under vegetal conditions which potentially captured the minimum and maximum distance over which a covey-call could be detected. Because brush density can influence the distance sound travels, we selected listening posts (i.e., study plots) representing low-brush density (6%) and high-brush density (30%). In order to select these areas we conducted an unsupervised classification of a 1-m resolution color aerial photograph using ERDAS IMAGINE 7.0 (Leica Geosystems, Heerbrugg, Switzerland). We used woody cover, herbaceous cover, bare ground, water, and shadow as our classes. We then visually selected a point that was centered in an open area with little woody cover (low-brush density) and one that was centered in dense brush (high-brush density). The point selected was where the observers would listen for calls. Vegetation directly between the observer and the sound source has a greater effect on the distance a call is heard than the surrounding vegetation (L. Hewett, Physics Department, Texas A&M University-Kingsville, personal communication); therefore, we clipped strips from the classified image that were 40 m wide and 1300 m long radiating out in the 4 cardinal directions from these points (Figure 1). These strips corresponded to the corridor that sound would travel to reach the observers. We then calculated canopy coverage of woody plants within these 4, 40 x 1,300-m strips. The low-brush density study plot contained 6% brush canopy coverage, and the high-brush density study plot contained 30% brush canopy coverage.

AUDIBILITY TRIALS - We conducted audibility trials using a randomized complete block design where brush-density types (n = 2) were the experimental units and observers were the blocks (n = 4). We conducted audibility trials during November, 2005 on days with little to no breeze. Audibility trials involved 4 trained observers and 4 recording-playback assistants, which remained consistent throughout our experiment. We began each trial approximately 2 hours before sunrise to mimic

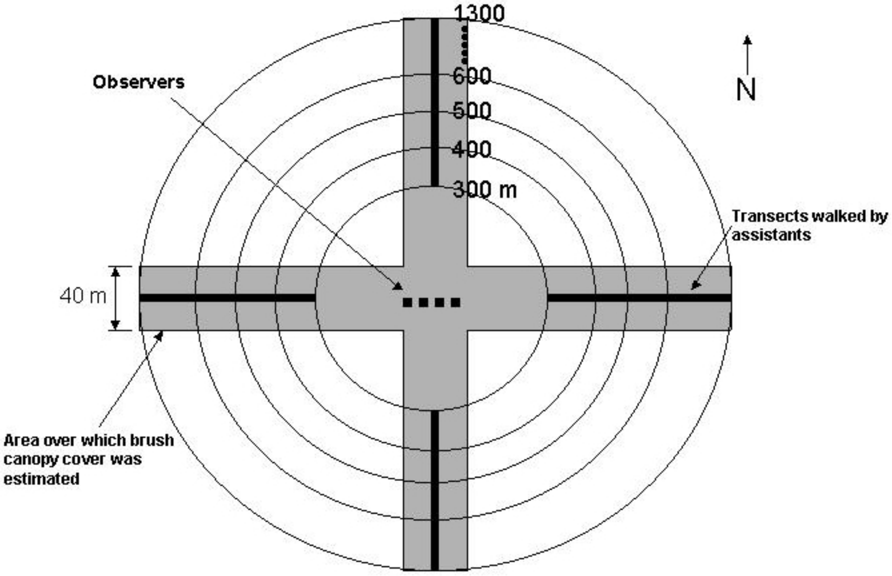

Figure 1: Schematic of experimental layout to test radius of audibility. Observers were positioned in center, oriented north, and spaced 10 m apart. Assistants played calls in the 4 cardinal directions at 100 m intervals from 300 m up to 1300 m.

conditions under which covey-call surveys are normally conducted but to complete the trial before wild birds initiated their calling. Observers were aligned along an east-west transect, oriented northward, and separated 10 m apart in order to isolate observers and prevent intra-observer influence (Figure 1). Recording-playback assistants then moved away from observers in each of the 4 cardinal directions and played the recording of the covey-call every 100 m. Recordings were played using 4 CD radio cassette recorders (model CFD-S350, Sony Corporation of America, New York, New York), one for each assistant. Recordings were broadcasted at about 100 db, the decibel-reading characteristic of covey-calls that was determined earlier during our recording sessions. Play-back assistants positioned CD players such that speakers were about 15 cm off the ground, the approximate height of a bobwhite. Assistants communicated using two-way radios (Motorola TalkAbout T5500, Motorola Incorporated, Schaumburg, Illinois) to avoid playing the recording at the same time. Assistants recorded the number of times the recording was played and the time of playing. Observers were instructed to record the time they heard a covey-call, the direction of the call, and the number of calls. Observers and assistants had time-synchronized watches.

We compared data (e.g., number of times recordings played, recording time) between observers and assistants. From this comparison, we determined the maximum distance over which the recording was detected for each observer in each direction. Mean radius of audibility was simply the maximum detection distances averaged across the 4 directions for each observer. We compared radii of audibility between brush density types with ANOVA using a randomized, complete block model.

Observer Variability

We measured observer variability using a completely randomized design. Our experiment involved 3 listening posts (experimental repetitions) and 12 observers (experimental units) that were randomly assigned to a listening post. Prior to experiment, observers were trained on morning covey-call methodology and listened to a covey-call recording to further refine their ability to detect and identify

Table 1: Radius of audibility (m) of the morning covey-call played at 100 db at low (6%) and high (30%) brush densities in Kleberg County Texas, November 2005.

		Transect Direction					
Brush Coverage	Observer	N	S	E	W	Mean	SE
Low-brush density							
	1	500	1100	1000	400	750	175.59
	2	800	1300	1200	1000	1075	110.87
	3	800	1000	1200	900	975	85.39
	4	1000	1300	1200	600	1025	154.78
	Pooled					956	71.72
High-brush density							
	1	1200	900	1100	800	1000	91.29
	2	1100	900	800	700	875	85.39
	3	1000	1300	1200	800	1075	110.87
	4	700	900	800	700	775	47.87
	Pooled					931	66.44

the call. For observer-variability experiments, observers arrived at posts 45 minutes before sunrise and listened for covey calling. Observers recorded time of first and last call, number of calling coveys, and approximate location of each covey. Weather conditions (temperature, humidity, and wind speed) also were recorded during each experiment. We calculated mean number of coveys detected, range, and percent coefficient of variation (CV) for each experiment.

Calling Behavior

We documented probability of coveys calling and number of individuals per covey calling during October-December, 2004-2005. To do this, we located radio-marked coveys about 45 minutes before sunrise and homed within 20 m of the covey. At the onset of calling, we recorded whether or not the covey called, the number of birds that called within the covey, and the number of calling events. A calling event was defined as a series of covey-calls separated by >1 minute (Wellendorf et al. 2004). We calculated 95% confidence intervals for probability of coveys calling using ($p \pm Z$ 0.025 SE[p]) where p is the proportion of coveys calling and SE(p) is the square root of $[p(1-p)/n]$.

Results

We determined 100 db to be the appropriate sound level to play the recording. This was based on our most reliable sound meter reading obtained within 10 cm of a captive bird. The mean radius of audibility was similar between the low-brush (956 ± 72 m; mean ± SE) and high-brush density area (931 ± 66 m; P = 0.75; Table 1). Pooled over areas, mean radius of audibility was 944 ± 46 m. Because we played covey-call recordings only at 100-m intervals, we estimated an overall radius of audibility of 900 m to more accurately reflect the precision at which our estimates were measured. Using 900 m as the radius of audibility, we calculated that counts surveyed an area of 254 ha.

We measured considerable variation between observers while conducting the morning covey-call survey. Percent coefficient of variation ranged from 18% to 49% (Table 2). Regarding calling behavior, 67% ± 9.0 (n = 30 coveys) of coveys emitted the covey call during 2004. Of the 20 coveys that

Table 2: Observer variability in number of coveys detected during morning covey-call surveys for northern bobwhite in Brooks and Kleberg Counties, November 2005.

	Observer					
Experiment Repitition[a]	1	2	3	4	Mean	CV
1	14	9	8	8	9.75	29%
2	6	2	3	3	3.5	49%
3	9	6	9	9	8.25	18%

[a] Each repetition consisted of different observers

called, 14 (70% ± 10.2) involved more than 1 bird calling within the covey (Table 3). In addition, we documented an average of 2.6 ± 0.32 calling events/covey/morning. Calling events per covey were 1 calling event ($n = 5$ coveys), 2 (n = 6 coveys), 3 (n = 4 coveys), 4 (n = 3 coveys), 5 (n = 1 covey), and 6 (n = 1 covey). During the same time period in 2005, we documented 88% ± 7.8 (n = 17) of coveys emitting the covey-call. Of the 15 coveys that called, 14 (93% ± 6.4) involved more than 1 bird calling within the covey (Table 3). In addition, we documented an average of 3.2 ± 0.22 calling events/covey/morning. Calling events per covey were 1 calling event (n = 1 coveys), 2 (n = 1 coveys), 3 (n = 7 coveys), and 4 (n = 6 coveys).

Discussion

Our empirical estimate of radius of audibility (900 m) for vegetal conditions of South Texas was higher than any estimate used in prior research. Other studies have used radii of audibility consisting of 400 m (Roseberry 1982), 700 m (DeMaso et al. 1992), and 800 m (Guthery 1986). Other than DeMaso et al. (1992), who estimated radius of audibility based on physical laws of sound travel, these prior estimates represented arbitrary values. Although our estimate of radius of audibility was obtained using experimentation, we do not suggest that our estimate be used as the exact value. Factors such as wind, topography, and background noise affects the distance over which a sound travels. We documented that density of woody plants did not influence radii of audibility in our study. However, our study was conducted on level terrain with low growing brush (10-15 m). This relatively low height of woody plants permitted sound to travel just above the brush layer (L. Hewett, Physics Department, Texas A&M University-Kingsville, personal communication). In landscapes with taller vegetation, woody plants may impede the travel of sound and therefore impact radius of audibility. We recommend that the appropriate radius of audibility be determined specifically for each study area. Determining the appropriate radius of audibility is important because it determines survey area and therefore estimated density. The various radii assumed by prior research corresponded to survey areas which differed drastically: 50 ha (Roseberry 1982), 154 ha (DeMaso et al. 1992), and 200 ha (Guthery 1986). Naturally, this large impact of radius of audibility on survey area would affect the accuracy of this survey method. Point transects (Buckland et al. 2001) may be used as an alternative to fixed-radius estimators such as the morning-covey call method; however, accurate distance measurements to the calls (or placement into a correct distance interval) are required which may not be possible given variability in call pitch and loudness from the same

Table 3: Number of individuals emitting covey-call within a covey during morning covey-call surveys, Brooks County, Texas, October-December, 2004-2005.

			No. birds calling					
Year	Observation	n	1	2	3	4	5	Total birds calling
2004	Coveys Calling	20	6	12	2	0	0	36
2005	Coveys Calling	17	1	3	5	5	3	57

covey (see below).

We also obtained an empirical estimate of the loudness of the covey-call. Our estimate (100 db) corresponded to that of DeMaso (1991, 101 db), which was calculated based on the Calder (1990) equation. By assuming sound dissipated over a hemisphere and using the threshold of human hearing (10^{-12} watts/m^2), DeMaso (1991) calculated a radius of audibility of 15 km, which he realized was unrealistic. Alternatively, he speculated the loudness of the covey-call fell between 60 and 70 db thereby suggesting a radius of 700 m. Because we observed decibel readings of 100 db, Calder's equation predicting 101 db for a covey-call may not be unrealistic but rather that the assumptions of DeMaso (1991) (dissipation of sound over a hemisphere and threshold value of human hearing) are incorrect or unrealistic.

Prior research identified 2 primary limitations of the morning covey-call survey: observer variability and calling behavior. Both of these factors influence the number of coveys detected and therefore density. We documented that observers varied considerably in their estimate of coveys heard at a point. DeMaso et al. (1992) also reported observer variability as a limitation of the survey method. In their study, morning covey calls could not be used as an index to abundance for 1 of 3 observers. Wellendorf et al. (2004) suggested that differences among the ability of observers to count covey-calls should be considered and that observers should be trained to lessen the observer effects (Kepler and Scott 1981).

Calling behavior of bobwhites also complicates application of the survey. DeMaso et al. (1992) noted that morning covey-call surveys assumed that probability of calling was constant over space and time. Our and prior research indicate violation of this assumption. We documented probability of calling varied between years (20% difference). Seiler et al. (2002) reported that calling rates differed by 6% between years and fluctuated considerably (50-100%) within a year. Wellendorf et al. (2004) also documented substantial within year variation (48-87%) with a mean calling rate of 58%. In addition, we documented that more than one bird in a covey called. This finding is important because having multiple bobwhites emit the covey call could bias number of coveys detected if observers confused them as separate coveys. We documented an instance in which a covey with multiple calling individuals sounded as different coveys because individual birds were calling at different pitch levels and loudness (F. Hernndez, Caesar Kleberg Wildlife Research Institute, personal observation). Naturally, such additional variation would further bias morning covey-call surveys.

Management Implications

Although the morning covey-call survey may be used to obtain crude estimates of abundance, people employing the technique need to be fully aware of its limitations. Refinements to the survey method include using a 900-m radius of audibility (at least as an initial value) and correcting for probability of coveys calling (Wellendorf et al. 2004). Because the probability of calling varies annually, this variable should be estimated each year counts are conducted. In addition, observers should be kept constant from year to year as much as possible to reduce observer variability.

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Estimating Sample Sizes for Distance Sampling of Autumn Northern Bobwhite Calling Coveys

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Point transect sampling of calling coveys has been advocated for estimating autumn abundance of northern bobwhite (*Colinus virginianus*; hereafter bobwhite). We conducted power analysis, over a range of expected bobwhite calling covey densities to determine levels of sampling required to obtain density estimates for calling coveys over a wide range of precision. We used distance/detection information for autumn bobwhite coveys from 701 observer-mornings on 39 farms in the Upper Coastal Plain of Georgia to construct a global detection function (Uniform with cosine adjustment) using Program DISTANCE. We used simulation models to determine the expected coefficient of variation (CV) on density in relation to number of points sampled. We generated 1,000 sets of random samples in increments of 10 at sample sizes of 10-1,000. At each sample size we generated the respective number of observations from a Poisson distribution with λ = 0.5-3.0 and computed the density and associated statistics using the global detection function. We report the mean CV on covey density at each sample size. As expected, the CV on density decreased with increasing sample size and expected number of detections per point. Assuming sufficient observations to estimate the detection function, a CV on density $<$15% could be achieved with 50 points at densities with a mean detection of 1 covey/point or 20 points with a mean detection of 2 coveys/point. A mean CV $<$10% required 100 points at 1 covey/point and 30 points at 2 coveys/point. These simulations demonstrate that distance-based autumn covey surveys can provide density estimates for calling coveys with reasonable precision given sufficient effort.

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Key words: call counts, *Colinus virginianus*, density, distance sampling, Georgia, northern bobwhite, sample size, simulation

Introduction

Precise estimation of northern bobwhite (*Colinus virginianus*; hereafter bobwhite) population size is necessary to understand population dynamics, set population-based hunting regulations, and evaluate effectiveness of habitat management (Schwartz 1974, Stauffer 1993). Development of precise and unbiased techniques to estimate bobwhite population density, however, has constituted an enigma for biologists. Several approaches have been used to index (Bennett and Hendrickson 1938, Kozicky et al. 1956, DeMaso et al. 1992) or estimate (Dimmick et al. 1982, Guthery et al. 1988, Guthery and Shupe 1989, Janvrin et al. 1991) density of bobwhite populations. Point transect sampling of autumn bobwhite populations using covey calling activity has recently been advocated as a means to estimate bobwhite density (Seiler et al. 2002, Wellendorf et al. 2004, Wellendorf and Palmer 2005).

DeMaso et al. (1992) used autumn calling activity to index bobwhite density but reported poor correspondence between covey calling activity and density estimated via line-transect sampling (Guthery 1988). They attributed this lack of correspondence to variation in calling activity, proportion of coveys calling, and observer ability to differentiate coveys in high density areas. Recent studies (Seiler et al. 2002, Wellendorf et al. 2004) have established empirical relationships among calling activity, weather, and density; thereby addressing most concerns of DeMaso et al. (1992). When adjusted for calling rate, Wellendorf and Palmer (2005) reported similar

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covey density estimates of autumn bobwhite populations between point transect sampling and those estimated via quadrant surveys.

Point transect sampling of autumn bobwhites based on covey calling activity is less labor intensive than flush count (Guthery 1988, Janvrin et al. 1991) or mark-recapture (Dimmick et al. 1982, Guthery and Shupe 1989) techniques. However, point transect sampling of bobwhites is subject to several biologically-based constraints. Peak calling activity of coveys occurs during a relatively narrow window of time (2-3 weeks) during autumn and calling occurs most reliably only during a brief period (20 min) before sunrise (Seiler et al. 2002, Wellendorf et al. 2002, 2004); limiting data collection to 1 point/observer/morning. Given these constraints, sampling of multiple points at multiple sites within the brief window of peak calling activity becomes problematic. *A priori* power analysis to determine appropriate levels of sampling required to obtain density estimates with desired levels of precision will facilitate efficient use of resources. Furthermore, researchers and managers will be able to better evaluate the tradeoffs between sampling intensity and statistical power to detect treatment effects when developing sampling protocols to meet research or management objectives while minimizing superfluous sampling effort (Steidl et al. 1997). Our objectives were to use simulation models based on field data measurements of observer-covey distances to estimate sample sizes required to meet desired levels of precision on covey density using point-transect distance sampling of autumn bobwhites.

Study Area

Our study was conducted on 39 privately owned farms enrolled in the Georgia Department of Natural Resource's Bobwhite Quail Initiative (BQI) in the Upper Coastal Plain physiographic region of Georgia. The BQI consisted of 3 focus regions (East, Central, and Southwest) where state-sponsored cost-share incentives were offered for bobwhite habitat development (e.g., prescribed burning, field borders, and conservation tillage). Major land uses were intensive row crop (cotton, peanut, soybean, corn, and winter cereal) agriculture and timber production. However, agricultural intensity varied among the 3 regions, with mean cropland area of 31%, 19%, and 12% for the Southwest, East, and Central regions, respectively (http://www.georgiastats.uga.edu/, February 2003). Mean row crop field size of farms sampled in this study was approximately 23 ha. Forested areas were plantations of loblolly (*Pinus taeda*) and slash (*Pinus elliotti*) pine, with occasional stands of longleaf pine (*Pinus palustris*). Forested land comprised approximately 63% of the Central Region, 62% of the East Region, and 46% of the Southwest Region (http://www.georgiastats.uga.edu/, February 2003). For a complete study area description see Hamrick (2002).

Methods

Field Data Collection

Because distance sampling of autumn bobwhite populations is a new technique, few data sets of sufficient breadth to capture variability in landscape context and bobwhite density (i.e., not site-specific studies) and depth (i.e., number of detections) exist from which observer-covey distances could be used to generate a detection function. In lieu of this data, we generated *post hoc* observer-covey distances from quadrant surveys conducted as part of the population monitoring program of BQI.

Quadrant Approach

The quadrant survey entailed placing one observer at the midpoint along each side of a 25 ha (500 m x 500 m) sampling cell (Wellendorf and Palmer 2005, ; Figure 1). Observers listened for the assemble, or "koi-lee," call (Stoddard 1931) given by bobwhite coveys and recorded the time, azimuth, duration, estimated distance to the covey, and number of covey calls per calling event for coveys within and outside of the quadrant. Surveys ended at sunrise if no calls were detected and were not conducted during periods of sustained rainfall. Upon completion of the survey, observers compared measurements to determine the number and estimated location of coveys. For each covey that was detected by >1 observer, the intersection of azimuths to the covey was used to plot the estimated covey location. Each covey location estimate was plotted on a final field map.

To minimize observer bias, all observers were trained by listening to recorded covey-calls and by spending several mornings in the field listening to calling coveys pointed out by experienced observers before conducting covey-call surveys (Kepler and Scott 1981, Smith 1984, Scott et al. 1981, Seiler et al. 2005). We assumed that observers were able to determine the direction from which a calling covey was heard with reasonable accuracy. Seiler et al. (2005) estimated a mean measurement error of 75 m between known covey caller locations and paired azimuth-derived locations in rolling terrain. Other assumptions of distance sampling that we believed we met were that coveys were detected at their initial location and that all coveys calling at survey points were detected (Buckland et al. 2001). Whereas inter-observer variation in detection probability may substantially affect resulting density estimates (Diefenbach et al. 2003), we assumed no inter-observer variation in observer ability to detect calling coveys. This additional assumption was necessary because we did not have sufficient data to test differences in observer-specific detection functions. We believe this to be a reasonable assumption given that Wellendorf and Palmer (2005) reported observer-specific detection rates to be within 10-15% of the overall mean detection rate among trained individuals conducting autumn calling covey counts in Florida.

Observer-covey Distances

Multiple detections of the same covey permitted estimation of distances from observers to calling coveys. Distances from observers to covey locations were calculated by the intersection of azimuths for >1 observers that detected a particular calling covey. Most covey locations were estimated via the intersection of azimuths from 2 observers. If greater than 2 observers detected the same covey, we used the geometric center of the error polygon created from the intersection of all observer azimuths. Distance from the observer to the predicted covey location was estimated using standard trigonometric relationships between observer locations along the quadrant and reported azimuths to calling coveys (Figure 1).

Our intent was only to use the quadrant data to generate observer-covey distances for developing a detection function for use with simulated covey observation data; therefore, the experimental design (i.e., replication, repetitions, treatment, etc.), and subsequent density estimates, used in the BQI monitoring program were irrelevant and are not addressed. All distances were computed to the nearest 5-m increment. Wellendorf and Palmer (2005) reported that well trained observers could reasonably classify calling coveys into distance categories of 0-100 m, 101-250 m, 251-500 m, and >500 m. Therefore, to simulate probable point-transect distance sampling data, we grouped observer-covey distances into these respective distance categories. Right truncation was set to 700 m, an assumed mean maximum audible range of detection DeMaso et al. (1992).

Detection Function

Prior to analyses, we visually inspected the data by plotting observations by distance category to determine potential detection functions that would best fit observed data patterns. The uniform base function with cosine or hermite polynomial adjustment terms and the hazard rate base function with either cosine or simple polynomial adjustment terms were selected as likely base function-adjustment term combinations that would best model the data. We used Program DISTANCE (Thomas et al. 1998) to fit models and subsequently identify a detection function to estimate the detection probability ($h_{(o)}$), the value of the probability density function $f_{(x)}$ evaluated at 0. Base functions and series expansion terms, increasing in complexity (# of estimable parameters), were sequentially evaluated by comparing Akaike's Information Criterion (AIC) values

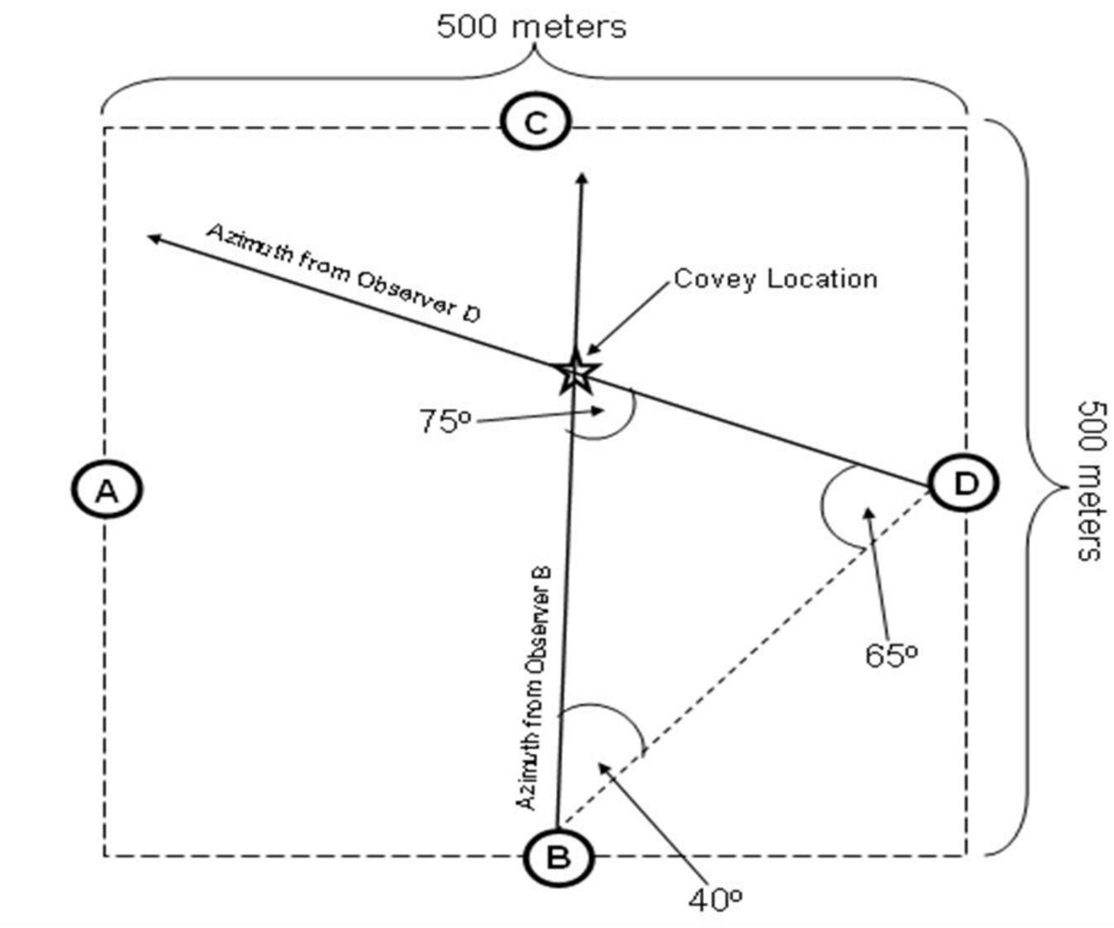

Figure 1: Depiction of how observer-azimuths to calling northern bobwhite coveys (taken during multiple-, independent-observer covey-call-count surveys) were utilized to calculate estimated observer-covey distances.

among competing models (Anderson et al. 2000, Buckland et al. 2001, Burnham and Anderson 2002). When a more complex model failed to adequately fit the data relative to the number of parameters within the model (greater AIC), the previous model was selected as the best approximating model (Buckland et al. 2001).

Simulations

To determine the expected effect of sample size (number of sampling points) on precision of density estimates (i.e., coefficient of variation; CV), we generated 1,000 sets of covey detections from a Poisson distribution in increments of 10 at each sample size from 10-100, in increments of 100 for sample sizes of 100-500, and at 1,000. Because bobwhite densities vary substantially across their range, we repeated this process for Poisson distribution means of 0.5, 1.0, 2.0, and 3.0 detections (i.e., calling coveys)/point. Using the detection probability ($h_{(o)}$) and standard error estimated from the best approximating detection function, we then computed the covey density and CV as described in Buckland et al. (2001) for each of the 1,000 samples at each sample size/mean covey detection combination. All simulations were conducted using programming statements in SAS statistical software (SAS Institute, Inc. 2002). We *a priori* set a CV of 15% as an acceptable

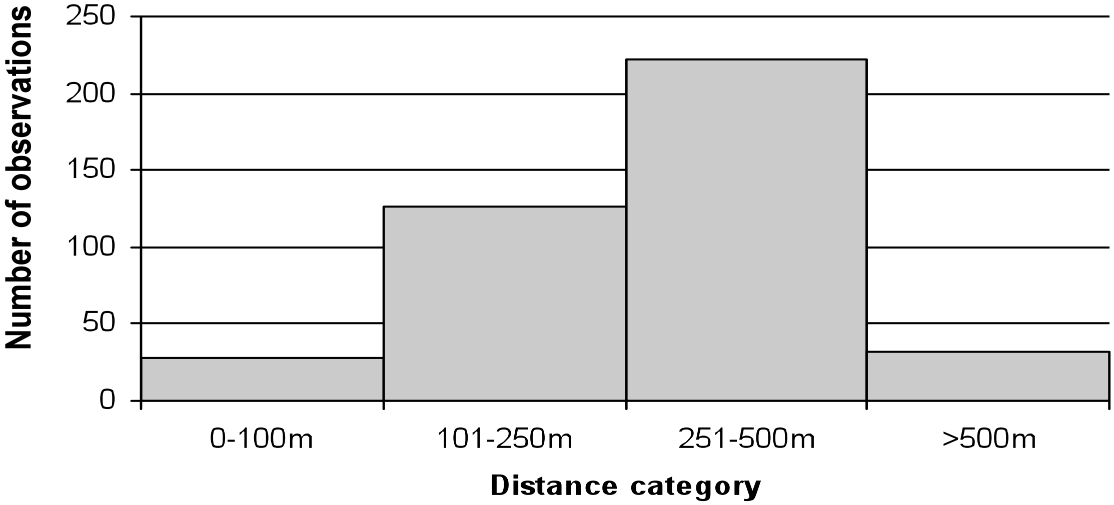

Figure 2: Observer-covey distances of calling northern bobwhite coveys computed via triangulation of azimuths to calling coveys taken by pairs of independent observers positioned 350 m and 500 m apart during quadrant sampling. Distances were then categorized into 0−100m, 101−250m, 251−500m, and 500m+ distance categories, Georgia, 1999−2000.

level of precision.

Results

We computed 408 observer-covey distances from 701 observer mornings for use in estimating a detection function and subsequent detection probability (Figure 2). The uniform base function with a cosine adjustment term (order 1) was selected as the best detection function model (AIC = 883.06) and fit the data well ($\chi 2^2 = 1.7034$, $P = 0.42669$). Effective detection radius was 381.72 m (SE = 4.977). Detection probability accounted for 10.9% of the variation of density whereas the encounter rate accounted for the balance of this variation.

As expected, the CV on density decreased with increasing sample size and expected number of detections per point (Figure 3). A CV <15% could be achieved with 50 points at densities with a mean 1 covey detected/point or 20 points with a mean detection of 2 coveys/point. A mean CV <10% required 100 points at 1 covey/point and 30 points at 3 coveys/point. Our simulations suggest that with a sample of 40 points a CV of 16.1% could be expected and with 50 points a CV of 14.5%. Population variability stabilized at a CV of approximately 4%.

Discussion

The availability of observer-covey distance data in agricultural landscapes of the southeastern United States is scant; we were only able to generate *ad hoc* observer-covey distances from previous research. Therefore, we acknowledge several potential biases in this analysis. First, we assumed that observers could reliably detect the direction of, and accurately measure an azimuth to, a calling covey. We further assumed and that these azimuths were recorded without error. From our experiences, and from those of others (Seiler et al. 2002, Wellendorf et al. 2004, Seiler et al. 2005), we do not consider this source of error extremely problematic and assume directional error, and subsequently estimated distance error, to be random. Obviously, as the distance from the observer increases, location error of the covey will increase. This error would affect the detection function, detection probability, and resulting standard error of the detection probability. Random errors in distance measurement, however, are tolerable if they are not too large, and sample size

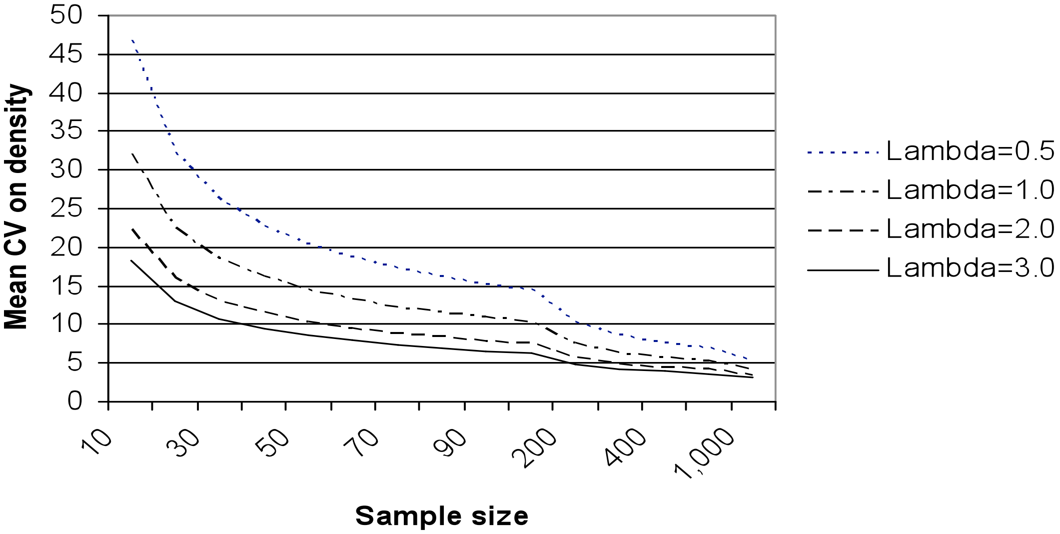

Figure 3: Mean coefficient of variation (CV) of density estimates generated from 1,000 samples (with replacement) for number of points sampled using estimated covey-observer distances from Bobwhite Quail Initiative sites in Georgia, 1999–2000.

is large (Buckland et al. 2001). Minor variations in the detection probability and standard error had little effect on the mean CV on density (M. Smith, unpublished data, Mississippi State University).

Landscape context, hence the structure and composition of land cover within the landscape, may substantially influence detection probability (Bibby et al. 1992, Buckland et al. 2001). The detection function (uniform with cosine adjustment) used in our study was constructed from covey observations obtained in agricultural landscapes within the Upper Coastal Plain physiographic region and may not be applicable to other physiographic regions or landscapes. Cropland varied from 12-31% in landscapes in which counts were conducted. Within forested landscapes under intense bobwhite management in north Florida, Wellendorf and Palmer (2005) reported use of a uniform base function with a simple polynomial adjustment, but model fit was marginal. Although the amount of forested area differed among our sites, we did not have sufficient data to test differences in detection function among the three areas. Given that all of the sites were relatively open agricultural lands, we suspect only minor, if any, differences in detection probability among sites. Similar to other distance-based techniques (Guthery et al. 1988), sampling in areas of low bobwhite densities will be problematic (Kuvlesky et al. 1989). First, sufficient numbers (approximately 70-100; Buckland et al. 2001) of detections may not be obtained to in order to estimate detection functions and secondly, variance will be exceedingly high such that confidence intervals on density will be rendered uninformative.

Management Implications

In October 2004, the Farm Services Agency of the U.S. Department of Agriculture (USDA) announced the availability of a field border practice (CP33-Habitat Buffers for Upland Birds) within the Continuous Conservation Reserve Program. Whereas CP33 is a USDA farm bill conservation practice, state wildlife agencies were delegated the responsibility of developing and implementing a statewide monitoring program that will 1) provide statistically valid estimates of bobwhite density (or some other appropriate measure) on fields enrolled in CP33 at state, regional, and national levels, and 2) provide a measure of the relative effect size of the CP33 practice at state, regional and national levels. The Research Committee of the Southeast Quail Study Group developed a national protocol (Burger et al. 2006) that states could use to meet these objectives. This national monitoring protocol outlined a suggested multi-stage sampling framework and infield protocol to ensure consistency in data collection among states and to facilitate statistically valid measures of the effectiveness of CP33. Point transect sampling was selected as the primary technique for monitoring breeding season bobwhites and songbirds and fall covey densities. Fourteen states adopted this protocol.

Given the statewide availability of the CP33 practice and the relatively limited resources of most state wildlife agencies to conduct monitoring, it was paramount to *a priori* determine an appropriate level of sampling (number of points surveyed) that would provide reasonable (CV $\leq$15%) estimates of bobwhite density while minimizing superfluous sampling effort. We used the approach outlined in this paper to estimate adequate sampling intensity at state, regional (BCR), and national levels. This simulation suggested that at a sample of 40 points a CV of 16.39 could be expected and at 50 points a CV of 14.69. From this simulation we concluded that 40 fields/state would produce estimates sufficiently precise to meet the language in FSA Notice CRP-479 *at the state level* and will produce CVs on regional and national data in the 5-6% range. If fields enrolled in CP-33 were paired with un-enrolled control fields in the vicinity of each contract we could estimate the effect size of the CP-33 practice (number of quail/ac added to the landscape as a result of CP-33) and extrapolate that to the national enrollment to produce a defensible estimate of the national effect of CP33 on bobwhite and select songbirds. The National CP33 Monitoring Protocol recommended that sampling intensity should vary in relation to the number of acres enrolled in the state (i.e., proportional stratified sampling). Under this scheme states would monitor from 40- 141 fields.

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Summer Male Call Index Relative to Nesting Chronology and Autumn Density of the Northern Bobwhite

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We studied breeding season male call counts and breeding behavior of the Northern Bobwhite (*Colinus virginianus*) to determine the relationship between male calling activity and nesting chronology. Additionally, we examined the relationship between breeding season call counts and fall population size. Standardized call count routes were conducted on 6 different sites located in southwest Georgia and north Florida during the breeding season months (1 Apr - 31 Sep) in 2001 and 2002. An information theoretic approach was used to evaluate a set of 7 candidate, linear-mixed models describing breeding season calling of bobwhite males. Of the candidate models, the model in which call counts depended on year and a quadratic effect of the number of incubating hens was the best approximating model, suggesting that the percentage of incubating hens had the greatest influence on activity of calling males. We also used multiple linear regression models to predict autumn northern bobwhite abundance from mean numbers of calling male bobwhites detected during the breeding season. Peaks in male calling activity occurring during June and July demonstrated a strong relationship (R^2 = 0.987) with autumn population size, suggesting breeding season call counts were useful indices of autumn bobwhite abundance.

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Introduction

The Northern Bobwhite (*Colinus virginianus*); hereafter bobwhite, uses a wide variety of vocalizations to communicate between coveys, among coveys, and individually. Stoddard (1931) defined these vocalizations based on group movement, food finding, avoidance of enemies, and reproduction. The purpose of the distinctive "bobwhite" call, from which the species receives its common name, is thought to serve both breeding and territorial functions (Stokes et al. 1994, Rosene 1969, Stoddard 1931, Brennan 1999, Guthery 2000); however, the utility of the call is less understood.

During the past 5 decades, the male call count index (the average number of males heard per stop) has been evaluated as a population monitoring technique on several occasions (Ellis and Thomas 1972, Hansen and Guthery 2001, Rosene 1969, Robel 1969, Speake and Haugen 1960, Wells and Sexon 1982). However, there is apparent disagreement as to what the male call count index measures. While some researchers have successfully used summer counts of calling males to index autumn population size (Curtis et al. 1989, Ellis and Thomas 1972, Rosene 1969, Wells and Sexon 1982), others have demonstrated the poor reliability of call counts as predictors of fall bobwhite abundance (Hansen and Guthery 2001, Norton et al. 1961). Rosene (1969) speculated that the number of whistling males was an accurate predictor of fall population size and suggested that the number of whistling males heard corresponded to the number of coveys in the fall. He further proposed that the number of whistling males forecasted the success of fall hunting (Rosene 1969). Curtis et al. (1989) reported a high correla-

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tion (r = 0.94) between male call counts and hunting success. They also found that call counts correlated well with the total number of bobwhites harvested (r = 0.89) in the fall. In contrast, Norton et al. (1961) criticized the use of whistle counts to predict fall populations and, upon meta-analysis of previous works, noted: "It must be concluded that the case for usefulness of numbers of whistling cocks in summer to estimate autumn populations is weak and that a better method is needed." Hansen and Guthery (2001) reported seasonal variation in breeding season calling between years and noted the violation of assumptions required to determine annual trends in population abundance. Speake and Haugen (1960) suggested that the bobwhite call index could be developed into a useful tool for estimating progress of the nesting season while reporting that bobwhite whistling intensity fluctuates with nesting activity and that sharp drops in the call index likely correspond to peaks in hatching. Furthermore, researchers in the southeastern U.S. have noticed seasonal peaks in whistling by males throughout the nesting season and have often assumed a correlation between male calling and nesting activity (W. E. Palmer, Tall Timbers Research Station and D. C. Sisson, Albany Quail Project, personal communication). Stauffer (1993) reported that a more controlled research effort was needed to appropriately understand exactly what call counts measure. Therefore, we studied the calling behavior of bobwhite males on multiple sites in southwest Georgia and north Florida to develop protocols that would lead to improved accuracy and repeatability of data on nesting activity and population abundance derived from call counts. The primary objective of this investigation was to determine what conclusions may be deduced from male whistling during the breeding season and whether male whistling is an accurate estimator of nesting activity and/or fall population size.

Study Area

The study was conducted on 4 private lands in south Georgia and 2 in north Florida. These included 2 study areas (4,858 ha and 8,097 ha) in Baker County, Georgia, 1 (3,644) in Laurens and Bleckley Counties, Georgia and 2 (Tall Timbers Research Station) (1,741 ha) in Leon County, Florida and 1 (1,134 ha) in Thomas and Grady counties, Florida. These sites are in the Upper Coastal Plain physiographic region and characterized by old field, pine forests with relatively low basal area that are intensively managed for bobwhites. Intensive management regimes typically included annual burning, seasonal disking, drum-chopping, mowing, supplemental feeding, and mammalian nest predator control (see (Yates et al. 1995, Sisson et al. 2000*b*,*a*)). Typical field management consisted of autumn and late winter disking to stimulate annual weed and arthropod production. As a result of these intense management regimes, these areas maintained wild bobwhite populations ranging from 1.48 birds/ha to >7.41 birds/ha.

Methods

During March and April of 2001 and 2002, we trapped wild bobwhites using standard, baited funnel traps (Stoddard 1931). We radio-tagged and monitored a minimum of 50 quail (30 females and 20 males) on all sites during 2001 and 2002. Bobwhites were outfitted with a pendant-style radio transmitter (6.4 g) equipped with an activity switch (Holohil Systems Ltd., Ontario, Canada and American Wildlife Enterprises, Monticello, Florida), leg banded, weighed, aged, and released at their capture site. Trapping, handling, and marking procedures were approved by the Auburn University Institutional Animal Care and Use Committee. Beginning April 1, radio-marked bobwhites were monitored ≥ 3 times weekly to determine onset of nesting activity and/or mortality. All bird locations were determined using the "homing" method (Stauffer 1993) and were recorded on aerial photographs. We estimated breeding season (1 Apr - 31 Sep) survival for each site using the Kaplan-Meier product limit method (Kaplan and Meier 1958, Pollock et al. 1989) on a weekly basis. To account for potential capture and radio effects, mortalities occurring within 1 week of radio attachment and release were censored.

Table 1: Descriptive statistics for response (calling males) and predictor variables used in models of breeding male Northern Bobwhite calling activity.

Parameter	n	Mean	SD	Minimum	Maximum
Calling males	260	3.330	2.940	0.000	10.420
Hatching chronology	260	1.260	1.860	0.000	14.000
Pooled survival	260	0.660	0.200	0.230	1.000
Incubating hens	260	0.170	0.160	0.000	0.710

Radios lost due to unknown reasons also were censored on the day following the last day of normal contact.

We assumed birds observed in the same location on 2 consecutive days to be nesting. We approached inactive hens and marked their location with flagging tape at a distance of 5-10 m and recorded the location on an aerial photograph. We determined the exact nest location and number of eggs when telemetry indicated that the incubating hen was away from the nest. Nests were monitored daily. A successful nest was defined as a nest that hatched ≥1 egg.

Population Indices

Breeding season call counts - We developed a protocol based on previous research of male calling behavior (Curtis et al. 1989, Ellis et al. 1969, Hansen and Guthery 2001, Rosene 1969, Wells and Sexon 1982) to ensure accurate counts and to mitigate the influence of weather (i.e. wind, fog, rain, and cloud cover) on whistling males. Counts of whistling males were conducted along standardized call count routes each week at 5-9 day intervals (1 April through 31 September) during 2001 and 2002. Male calls were counted during the first 2 hours after sunrise [the ”calling optimum”; (Hansen and Guthery 2001, Rosene 1969) on days when the wind velocity was ≤16 kmh and cloud cover was ≤75%. Call count routes were comprised of 12 listening points 0.81 km apart, evenly distributed throughout the study area. The observer stopped and listened for 5 minutes at each point and recorded the start time, the number of whistling males, and climate conditions such as wind speed and direction, cloud cover, and/or fog. The average number of males calling per route was that week's call index. The call count route was run backwards on alternating weeks to decrease bias of optimal calling time and listening point locale.

Autumn population abundance - We used covey call count indices (DeMaso et al. 1992, Seiler et al. 2002, Wellendorf et al. 2004) to evaluate fall bobwhite abundance during 2000-2002. Covey call count surveys were conducted from mid-October to late-November. We used quadrat-sampling and point-count techniques to estimate autumn bobwhite density. The quadrat technique used a 25-ha (500 m x 500 m) quadrat to survey calling coveys. A total of 4 observers were required, with 1 observer positioned along the midpoint of each quadrat side. Observers listened for the "koi-lee" covey calls (Stoddard 1931) given by bobwhites, almost always before sunrise, and recorded the unique number of calling coveys on aerial photographs. When covey calling ended, observers compared their results to determine if a particular covey detected was inside or outside of the quadrat. Coveys detected inside of quadrats were used to estimate density via quadrat-sampling estimators. Each unique covey was plotted on an aerial photograph and flushed using pointing dogs to estimate average covey size. The point-count technique required a single observer to record approximate covey locations. We estimated fall abun-

Table 2: Model selection statistics for candidate regression models used to test competing hypotheses of male northern bobwhite calling behavior.

Model	-2(Log-ℓ)	$AIC_c{}^a$	$\Delta\, QAIC_c^b$	W_i^c	K^d
Year, Inc. hens, Inc. hens^2	459.5	476.1	0.000	0.993	8
Year, Inc. hens, Inc. hens^2, Year*Inc. hens, Year*Inc. hens^2	465.0	485.9	9.800	0.007	10
Year, Hatch chronology	592.3	606.7	130.600	0.000	7
Year, Hatch chronology, Year*Hatch chronology	594.4	611	134.900	0.000	8
Year, Pooled survival	612.3	626.7	150.600	0.000	7
Year, Pooled survival, Year*Pooled survival	614.3	630.8	154.800	0.000	8
Year-only	623.2	635.5	159.400	0.000	6

[a] Akaike's Information Criteria with a small sample bias adjustment.

[b] Akaike weight of evidence that the given model in the set of candidate models is the best model.

[c] Total number of model parameters (includes parameters for estimating intercept and variance).

[d] K is the number of parameters.

dance using point counts via the fixed-radius approach (Wellendorf et al. 2004). Covey-call surveys were adjusted via calling rate estimates (Wellendorf et al. 2004). Additionally, covey-call points and quadrats were randomly distributed across the study areas. Five sites were surveyed with quadrats. The remaining site was surveyed with point counts due to limited numbers of observers.

Analyses

Call count and breeding parameters - We used linear-mixed models [PROC MIXED procedure of SAS ®software (SAS Institute, Inc. 2006)] to estimate effects of site, year, weekly periods during the breeding season, hatching chronology, pooled survival, and incubating hens on mean numbers of calling male bobwhites. Hatching chronology was defined as the number of successful nests that hatched during a given week. The "incubating hens" predictor was defined as the number of hens incubating a nest divided by the number of hens alive (i.e., the proportion of hens incubating a nest). "Pooled survival" was the survival of males and females estimated by the Kaplan-Meier product limit method (Kaplan and Meier 1958, Pollock et al. 1989). There were 6 different sites where data were collected (4 sites in both 2001 and 2002, 1 site in 2001, and 1 site in 2002). Site was treated as a random effect (Littell et al. 1996). Counts of calling males were made for 26 consecutive weeks, April through October, during the breeding seasons of 2001 and 2002. This weekly time trend variable was treated as a random effect nested within sites and years with an autoregressive error structure. An autoregressive error structure was used because we assumed that calling activity among adjacent weekly periods would be more similar than calling activity at weekly periods farther apart (Littell et al. 1996). The predictor variable year was coded as an indicator variable (year 2002 was the baseline year for comparison with 2001) and considered a fixed effect. The fixed effects predictors hatching chronology, pooled survival, incubating hens, and the response variable mean numbers of calling male bobwhites were continuous variables (Table 1). For our analysis, we were most interested in how hatching chronology, pooled survival, and incubating hens affected male bobwhite calling

Table 3: Model coefficients for effects of numbers of incubating hens on calling male.

			95% CI	
Model Parameter	Coefficient	SE	Lower	Upper
Intercept	1.270	0.350	0.580	1.960
Incubating hens	0.660	0.040	0.570	0.740
Incubating hens^2	-0.240	0.030	-0.300	-0.180
Year	-0.360	0.100	-0.570	-0.160

activity during the breeding season.

Evaluation of scatter plots suggested that calling activity had a curvilinear relationship with incubating hens. To facilitate interpretation of regression coefficients, the continuous predictors and the response variable were standardized by unit normal scaling (Montgomery and Peck 1992).

An information-theoretic approach (Anderson et al. 2000, Burnham and Anderson 2002) was used to evaluate a set of 7 candidate models describing breeding season calling of bobwhite males in our study areas. Week and site effects were included in all candidate models. The year variable was also included in all candidate models to control for variation among years. We developed 2 nested models with hatching chronology, 2 nested models with pooled survival, 2 nested models with incubating hens, and a model with year effects only (Table 2). Though some models were nested, the candidate set of models was generally non-nested. The best approximating model in the set of candidate models was determined by Akaike's Information Criteria, adjusted for small sample bias (AIC_c) (Burnham and Anderson 2002). Model likelihoods computed from the PROC MIXED procedure were used to compute AICc and used to compare each candidate model. The model with the lowest AIC_c value was considered to be the best approximating model given the data. The relative plausibility of each model in the set of candidate models was assessed by Akaike weights (W_i) (Anderson et al. 2000, Burnham and Anderson 2002), where the best approximating model in the candidate set has the greatest Akaike weight.

Goodness-of-fit for the most highly parameterized model evaluating effects of hatching chronology, pooled survival, and incubating hens on calling activity was evaluated by residual analysis. Normal probability plots were constructed to evaluate whether serious deviations from normality existed for the most highly parameterized models evaluating effects of hatching chronology, pooled survival, and incubating hens on calling activity. Model residuals were plotted against predicted values and the continuous predictor variables to assess model fit. We further evaluated model adequacy by examination of model mean squared error (MSE).

Call count and autumn abundance - We used multiple linear regression models [PROC REG procedure of SAS ®software (SAS Institute, Inc. 2006)] to predict autumn bobwhite abundance from mean numbers of calling male bobwhites detected during the breeding season. We also controlled for potential variation among sites and years by including these terms in our models. Generally, there were at least 2 relatively high peaks in breeding season calling activity at all sites (see Results), and we used these 2 peaks to separately predict autumn northern bobwhite abundance (Table 4). For the 6 sites, Tall Timbers was coded as the baseline site for comparison

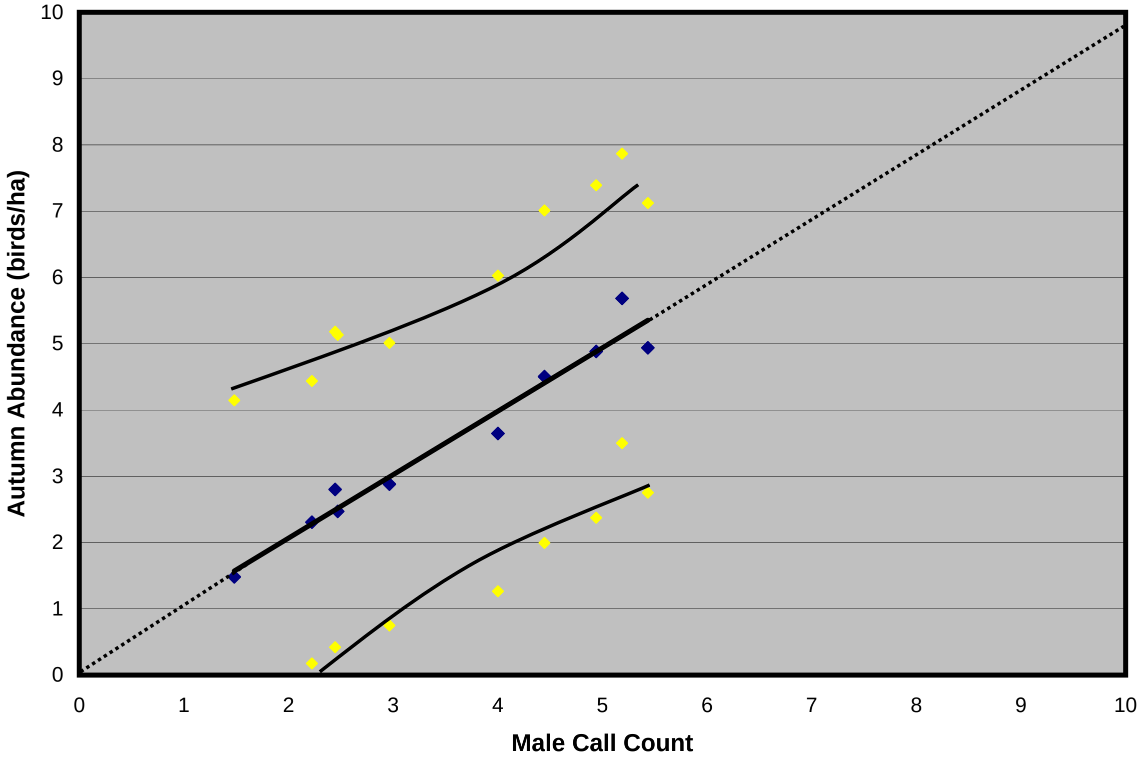

Figure 1: Regression model using the second peak in male calling with plotted prediction limits and confidence intervals for all study sites in Georgia and north Florida, 2001-2002. Dotted line denotes predicted abundance relative to predicted mean male call counts. Regression equation and coefficients: y = 0.20x + 0.22(yr) + 0.23(PBT) + 0.38(PBC) - 1.02(WTHL) + 0.20(PH) + 0.24(CP); $R^2 = 0.9867$.

with each of the other 5 sites. Year 2002 was coded as the baseline year for comparison with 2001.

Model fit was assessed by model coefficient of multiple determination (R^2) and mean squared error (MSE). Residual analysis was relatively uninformative because there were only 10 observations in each data set. During initial model fitting, no-intercept models of bobwhite abundance predicted from counts of breeding, calling males were found to best fit the data.

Results

Initial whistling of bobwhites occurred in early-April and continued into September. We observed as many as 3 peaks and as few as 1 peak in male calling and nesting activity dependent on year and site; however, generally 1 or 2 peaks occurred for most sites and years. The first peak was fairly consistent and occurred during the 7-9 week periods (mid- to late-May) while timing of the second peak was more variable and occurred during the 11-17 week periods (mid-June to late-July). The first peak yielded more intense calling activity than the second peak; although, the second peak in calling yielded a higher correlation to autumn abundance than the first peak.

Call Count and Breeding Parameters

The examination of residual plots suggested the fit for the most highly parameterized models evaluating effects of hatching chronology, pooled survival, and incubating hens on male calling activity was acceptable. Normal probability plots revealed some slight departure from normality for all of the models, but this departure did not appear to be severe. Estimated MSE for the most highly parameterized model with: 1) hatching chronology was 0.47;

2) pooled survival was 0.49; and 3) incubating hens was 0.26. Based on these model fit diagnostics, we assumed that the fit of the most highly parameterized models evaluating effects of hatching chronology, pooled survival, and incubating hens was acceptable, and the fit of subsequent candidate models also was adequate.

Based on model weights, the year, incubating $hens^2$ model suggested that the percentage of incubating hens had the greatest influence on activity of calling males (Table 2). There was virtually no support for any of the remaining candidate models based on model weights (Table 2). The year, incubating $hens^2$ suggested that calling activity was low when the percentage of incubating $hens^2$ was low to moderate, but calling activity was greatest when percentage of incubating hens was high (Table 3). Based on model selection criteria, pooled survival and hatching chronology were both poor predictors of breeding male calling activity compared to percentage of incubating hens.

Call Count and Autumn Abundance

Autumn bobwhite abundance demonstrated a strong, positive relationship with the mean number of calling males during both the first (MSE = 0.162; R^2 = 0.975) and second peaks (MSE = 0.105; R^2 = 0.987) in breeding season calling (Tables 5 and 6, Figure 1).

Discussion

Documentation of call count initiation and duration vary considerably depending on the study, species, and researcher (Elder 1956, Rosene 1957, Smith and Gallizioli 1965, Robel 1969, Brown et al. 1978, Hansen and Guthery 2001). Our results indicated that the initiation and duration of the call counts are critical to ensuring that peak calling is observed. Several researchers' protocols likely missed some of the valuable calling and nesting peaks we observed, due to late initiation dates and/or early cessation dates of call count surveys (Elder 1956, Rosene 1957, Smith and Gallizioli 1965, Robel 1969, Brown et al. 1978, e.g.). We found a positive relationship between male call counts and nesting activity; however, calling activity seemed to be more a function of within-season timing and nest success. Bennett (1951), Robel (1969), Kabat and Thompson (1963), Hartowicz (1964), Hansen and Guthery (2001), also reported peaks from mid-June to mid-July. Therefore, in contrast to that reported by Rosene (1957), the predictive capability of the male call index should not be based on a random 2-day sampling period; rather, the call count index should be implemented weekly for 6 to 8 weeks (1 Jun to 31 Jul) to determine the peak of calling activity. Additionally, our results suggest the peak occurring in June/July (second peak) was more descriptive of nesting activity, and purported nest success, and therefore may more accurately predict fall population levels. We note that Robel (1969) and Hansen and Guthery (2001) reported peak calling as late as August. These peaks were consistent with our findings; however, they were secondary peaks of smaller magnitude. The timing of these later, smaller peaks may be attributed to annual weather variations as reported by Hansen and Guthery (2001) or may be the result of asynchronous hatching caused by predation of nests and subsequent re-nesting attempts later in the breeding season. This latter pattern is often observed during the nesting season of the northern bobwhite resulting in the well known ”late hatch”.

Speake and Haugen (1960) concluded that bobwhite whistling activity fluctuates with nesting activity and peaks in hatching are preceded by sharp declines in calling. Robel (1969) reported hatching peaks 1 to 2 weeks following peaks of calling. We observed similar patterns in our data where, on several sites, increased hatching followed sharp declines in calling. Furthermore, when peaks in calling and nesting activity continued for several consecutive (3-4) weeks, hatching peaks were less pronounced and distributed over multiple (3-5) weeks. Robel (1969) reported similar results where in one year the whistling peak was short and sharply defined, while the next year the peak was longer and less distinct. Therefore, we surmised that sharp declines in calling and nesting activity indicate an increase in hatch-

Table 4: Descriptive statistics for response (autumn density) and predictor variables used in models of autumn northern bobwhite population density.

Parameter	n	Mean	SD	Minimum	Maximum
Autumn density	10	1.440	0.570	0.600	2.200
Calling males from first calling peak	10	7.080	3.070	1.580	10.420
Calling males from second calling peak	10	6.160	2.630	2.420	9.200

Table 5: Coefficients for regression model of autumn northern bobwhite density predicted from the first peak of breeding season counts of calling male northern bobwhites.

			95% CI	
Model Parameter	Coefficient	SE	Lower	Upper
Calling males from first calling peak	0.240	0.070	0.100	0.380
Site				
CP	-0.100	0.640	-1.350	1.150
PBT	-0.070	0.570	-1.190	1.050
PBC	0.140	0.350	-0.550	0.830
WTHL	-1.520	0.740	-2.970	-0.070
PH	0.420	0.480	-0.520	1.360
Year	-0.290	0.390	-1.050	0.470

ing activity (i.e., successful nesting) and less defined, more prolonged peaks indicate poor nest success.

Bennett (1951), Rosene (1957, 1969) suggested that the whistling male index is a reliable predictor of autumn populations. Conversely, Norton et al. (1961) indicated that such an index cannot accurately predict autumn population size because it lacks information regarding reproductive success and seasonal survival. This criticism appears unjustified because our study found that 2 peaks in breeding season calling were positively related to autumn population size. In fact, when using the second peak in breeding season calling, we observed a very strong relationship (R^2 = 0.9867) between breeding season call indices and autumn population size. These models indicate that the first and second peaks in breeding season call counts were useful predictors of autumn bobwhite abundance in 2 consecutive years at our study sites. However, it is unclear how much these relationships could change outside of year and site in this study. Therefore, the temporal and spatial context must be considered when using the call count index as a management tool for forecasting autumn population levels and subsequently establishing harvest rates.

Management Implications

This study demonstrates that the summer whistling male index of the bobwhite is a potentially useful indicator of breeding season progress

Table 6: Coefficients for regression model of autumn northern bobwhite density predicted from the second peak of breeding season counts of calling male northern bobwhites.

			95% CI	
Model Parameter	Coefficient	SE	Lower	Upper
Calling males from second calling peak	0.200	0.050	0.110	0.290
Site				
CP	0.240	0.430	-0.610	1.080
PBT	0.230	0.390	-0.540	0.990
PBC	0.380	0.260	-0.130	0.890
WTHL	-1.020	0.490	-1.980	-0.060
PH	0.200	0.380	-0.550	0.950
Year	0.220	0.240	-0.240	0.680

and autumn population levels when applied correctly. However, predictive power of the call count on fall population levels should be approached with caution as nest survival and brood survival are not accounted for in the breeding season call count index and tend to fluctuate among years depending on extrinsic factors such as weather and predator dynamics. To ensure accuracy and reliability of the call count, it is imperative to adhere to rigorous protocols to mitigate the effects of weather, site, and seasonal variation. We recommend conducting call counts weekly over a 2-month period (1 June - 31 July), rather than randomly selecting days or weeks within the breeding season, in order to ascertain the peak of calling activity (i.e., peak nesting activity). Call counts should be conducted during the optimal calling hours, within 1 hour after sunrise (Hansen and Guthery 2001), and performed under specific weather conditions as described previously in the ”Methods” section. Further study is needed to determine whether the 6- to 8-week monitoring period, as suggested in this paper, is an adequate measure of nesting activity and whether the peak observed during this period is correlated with fall population size. Our study was conducted on sites where bobwhite populations are stable to slowly increasing, and more research is warranted to determine whether these techniques are valid on low-density sites exhibiting rapid population growth or declines.

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Brood Ecology

Invertebrate Biomass and Richness in Various Food Plot Types in East Texas

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As northern bobwhite (*Colinus virginianus*) chicks are dependent on invertebrates for food, land managers often use spring/summer food plots to meet these needs. We examined invertebrate production in native vegetation and 6 different food plot types (i.e., fallow disking only; fallow disking and fertilizing; or disking, fertilizing, and planting a single species [browntop millet, iron and clay peas, or sorghum] or a multi-species mix [browntop millet, catjang peas, iron and clay peas, Japanese millet, and pearl millet]) in the Pineywoods of east Texas. Invertebrates were collected weekly during the summers of 1997 and 1999 and for 5 weekly sampling periods during summer, 1998. For each food plot type, invertebrates were separated from debris, air dried, and weighed as a group. Bi-weekly, a 100-invertebrate sub-sample was randomly selected from each sample and sorted to order with weight and number of individuals recorded. When spring precipitation was sufficient, multi-species food plots produced greater ($P < 0.05$) invertebrate biomass than fallow or native vegetation plots, and all cultivated plots had more ($P < 0.05$) biomass than native vegetation. Likewise, all cultivated plots had more ($P < 0.05$) biomass than fallow plots in early summer but not in mid- and late summer. A combination of multi-species (with legumes) food plots and fallow disking should provide bobwhite chicks with invertebrates throughout most summers.

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Introduction

Invertebrates are a critical component of the diet of northern bobwhites (*Colinus virginianus*; hereafter bobwhite). For hens, insects are a source of protein which is necessary for laying (Rosene 1969). More importantly, bobwhite chicks need 28% protein during the first 10 weeks of life (Rosene 1969) and are dependent on invertebrates for this protein (Handley 1931). As with most species, juvenile recruitment is critical in maintaining or increasing bobwhite populations. However, in the southeastern United States, recruitment has not replaced mortality for many years. In attempts to improve recruitment, managers and biologists use prescribed burning, fallow disking, and spring/summer food plots to increase invertebrate abundance.

In managing for bobwhites, fallow disking has long been recommended (Stoddard 1931). However, while disking may improve the structure of brood habitat (Olinde 2000), its impact, as compared to undisked native vegetation, on invertebrate production is unknown. Although disking is less expensive than planting food plots (Stoddard 1931, Brennan et al. 2000), food plots provide greater invertebrate biomass than native vegetation, either disked (Brennan et al. 2000) or undisked (Parsons et al. 2000*a*).

Some research has investigated invertebrate communities in food plots established using various combinations of fertilizer and plant species. In Mississippi, there were no differences in invertebrate density or biomass among old field (i.e., 2-year-old rye grass), fertilized old field, and fertilized Kobe lespedeza treatments; some invertebrate orders did differ among treatments, however (Jackson et al. 1987). In Georgia, invertebrate

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biomass did not differ among millet, sorghum, soybean, and wheat food plots, but invertebrate density was highest in millet and lowest in soybean and wheat plots. However, both biomass and density of Coleoptera, Diptera, Hemiptera, and Homoptera did differ among crop types; biomass and density values were generally highest in the millet plots (Maidens and Carroll 2002).

Burger et al. (1993) investigated invertebrate abundance, biomass, and diversity in Conservation Reserve Program (CRP) fields planted to 6 different species or species groups and in conventionally tilled soybean fields. They also tracked changes in the invertebrate community during 4 sampling periods between 1 July and 22 August. With few exceptions, invertebrate abundance, biomass, and diversity were highest in red clover fields and lowest in soybean fields. Likewise, these invertebrate values were generally higher in early July than mid-August.

No studies such as these have taken place in the Pineywoods of east Texas. Likewise, no studies have tracked changes in invertebrate biomass in food plots across the spring/summer brood-rearing period. Our objectives were to examine invertebrate abundance, biomass, and richness in native vegetation and in 6 types of food plots during late spring and summer, 1997, 1998, and 1999. The food plots were established by fallow disking only; fallow disking and fertilizing; or disking, fertilizing, and planting with browntop millet, iron and clay peas, sorghum, or a multiple species mix (browntop millet, Japanese millet, pearl millet, catjang peas, and iron and clay peas). For the purpose of this study, native vegetation was considered a food plot type.

Study Area

This study was conducted in the Pineywoods Ecological Region of east Texas (Gould 1975). The study area was on the 4,860-ha South Boggy Slough Hunting and Fishing Club. The club was on lands owned by Temple-Inland Forest Products Corporation and was approximately 16 km southwest of Lufkin, Texas. In 1989, company biologists selected 607 ha within the club to serve as a quail management area (QMA); this research project and several others (e.g., Nedbal et al. 1997, Liu et al. 2000, Parsons et al. 2000*a*) were conducted on the QMA.

Habitat modifications to favor bobwhites took place in the spring of 1989. Basal areas of the mature mixed pine-hardwood forests were reduced from 21-28 m^2/ha to approximately 14 m^2/ha. Sixty-nine food plots, ranging from 0.8 to 2.0 ha in size and totaling 81 ha, were established in the area. Beginning in 1989, approximately 60% of the QMA was burned annually; the remainder was burned biannually. A detailed description of habitat modifications on the QMA may be found in Dietz (1999).

Methods

Field Procedures

In January 1997, we selected 5 blocks on the QMA. Average distance between blocks was approximately 360 m; the shortest distance between any 2 blocks was 200 m. Although all blocks were on upland portions of the QMA, all were on soils classified as wet and/or clayey (Liu 1995).

Within each block, we established 6 cultivated food plots (0.8-2.0 ha in size) and 1 uncultivated food plot in native vegetation. We then randomly assigned 1 of the 6 cultivated food plot types to each food plot. Each spring, the food plots received the assigned treatment as early as possible, usually in early April. In order to maintain plant species homogeneity, each plot received the same treatment each spring. One food plot was fallow disked only; the remainder were disked and fertilized with 13-13-13 at a rate of approximately 220 kg/ha. Single-species plots of browntop millet, iron and clay pea, and sorghum plots were planted at rates of 45, 112, and 32 kg/ha, respectively. In the multi-species plot, browntop millet, catjang pea, iron and clay pea, Japanese millet, and pearl millet were planted at rates of 22, 22, 100, 28, and 22 kg/ha, respectively.

On the QMA, the bobwhite nesting period extended from mid-May into September. Most (70%) nests were initiated in May and June, thus most clutches of eggs hatched in June and July. However, nest initiation extended into September and at least 1 clutch hatched in October (Parsons et al. 2000*b*). Therefore, if the food plots had adequate vegetation, we began invertebrate sampling in mid-June and continued to do so on a weekly basis through early September. We used a gasoline-powered backpack vacuum machine to collect invertebrates (Ault and Stormer 1983). Starting at a random point, the operator moved in a zig-zag pattern through the food plot. We used the zig-zag pattern rather than designated transects to avoid sampling previously trampled vegetation. As the operator walked, the collecting cone was moved in a side-to-side motion within 15 cm of the soil surface; each food plot was vacuumed for 40 seconds.

Invertebrates and debris gathered in the food plot were immediately transferred to a labeled, self-sealing plastic bag containing an alcohol-soaked cotton ball. In 1997, we began sampling invertebrates on 26 June and gathered 11 weekly samples, ending on 3 September. Although food plots were planted in a timely manner, above-average precipitation in early spring (Table 1) made some plots inaccessible for sampling until late June. In 1998, there was a severe spring and summer drought and the cultivated food plots lacked vegetation until late August, thus we did not sample invertebrates in such plots that summer; invertebrates were collected in native vegetation during July and late August that summer. In 1999, we obtained 12 weekly samples, beginning on 21 June and ending on 14 September; due to equipment problems we were unable to collect invertebrates during the fourth week of July.

Laboratory Procedures

Samples were frozen at 0 °C for at least 48 hours to ensure that all invertebrates were dead. Thereafter, each bag was opened and its contents allowed to air dry for up to 36 hours. When dry, the contents were poured into a number 35 sieve. Obvious debris was carefully searched for invertebrates which were placed in a labeled vial; the debris was discarded. With the aid of a microscope, the contents remaining in the sieve were searched and invertebrates gathered. In doing this, soil particles fell through the sieve, making invertebrates easier to recognize. If necessary, the sieve was gently shaken to expose invertebrates; this was usually unnecessary. Searching continued until all invertebrates $\geq$0.5 mm in length were gathered and placed in the vial.

Contents of each vial were placed in a Petri dish and allowed to further air dry for 24 hours. The sample was then weighed to the nearest 0.0001 g. In order to evaluate taxon richness, samples collected on alternate weeks were sub-sampled to identify invertebrates to order. Contents of each Petri dish were evenly spread on a transparent plastic board with a 400-intersection grid etched on it. One hundred intersections were randomly chosen and the invertebrate nearest each selected intersection was gathered. The sub-sample was weighed and the invertebrates within it were identified to order; the weight and number of individuals of each order were recorded. If a sample contained fewer than 100 invertebrates, all individuals were sorted and weighed.

Statistical Procedures

We compared invertebrate biomass collected in native vegetation during July and August among the 3 years using a 2-way (week*year) univariate analysis of variance (ANOVA) and a Tukey test (Zar 1999, SAS Institute, Inc. 2006). We used a univariate ANOVA to determine if invertebrate biomass from 1997 and 1999 could be pooled. Biomass differed between years, so we used a 2-way ANOVA with a Tukey test to examine differences among food plots and among weeks for each year. When tests indicated differences, we used univariate ANOVAs with Tukey tests by week or food plot type to separate means. Invertebrate richness data were examined using a 2-way ANOVA and Tukey test of differences among food plots and weeks for each year. Finally, all biomass data were square root transformed and richness data log transformed (original values are reported) before analyses, with $\alpha = 0.05$.

Table 1: Precipitation (cm) at Lufkin, Texas, approximately 20 km northeast of the Quail Management Area in the Pineywoods of east Texas (National Oceanic and Atmospheric Administration 2006).

	Month								
Year	Jan	Feb	Mar	Apr	May	Jun	Jul	Aug	Total
1997	8.18	17.37	13.16	15.29	8.59	8.48	5.16	5.94	82.17
1998	22.99	13.74	6.86	7.47	0.03	2.03	5.11	12.34	70.57
1999	21.21	1.17	12.12	4.14	21.87	16.28	8.94	0.43	86.16
30-year $\bar{X}$	11.30	8.05	8.97	7.95	13.44	10.62	6.60	7.82	74.75

Results

Invertebrate Biomass

We identified 16 invertebrate taxa as potential food items. Araneida and insect orders comprised >90% of the total biomass in 1997 and 1999. The orders Hemiptera and Homoptera comprised >50% of the biomass each year. Only in native vegetation did these taxa not comprise the majority of the biomass (Table 2).

When we compared invertebrate biomass collected in native vegetation during July and August, 1997, 1998, and 1999, there was a week*year interaction ($F_{5,59} = 3.35$, $P < 0.001$; Figure 1); biomass differed among years ($F_{2,59} = 38.37$, $P < 0.001$) but not weeks ($F_{4,59} = 1.31$, $P = 0.278$). Mean weekly biomass in native vegetation in 1997 (0.1804 g) was higher than in 1998 (0.0498 g) or 1999 (0.07654 g). However, although differences were not significant, 1997 and 1999 biomass declined from 9 July values of 0.2094 g and 0.1215 g, respectively, to 29 August values of 0.1193 g and 0.0199 g, respectively. Conversely, in 1998, values on those dates increased from 0.0186 g to 0.0884 g.

Biomass of invertebrates collected in 1997 differed from that collected in 1999 ($F_{1,749} = 40.51$, $P < 0.001$). For 1997 data, there was no week ($F_{10,297} = 1.04$, $P = 0.407$) nor week*food plot interaction ($F_{60,297} = 0.76$, $P = 0.901$; Figures 2a, b). However, there were differences among food plot types ($F_{60,297} = 18.71$, $P < 0.001$). Overall, multi-species food plots produced more invertebrate biomass than did native vegetation, fallow, or fallow with fertilizer food plots (Table 3). However, invertebrate biomass differed among food plot types for only 4 weekly samples. Multi-species food plots produced more invertebrate biomass than either native vegetation or fallow plots in late June. By late July, biomass produced by fallow plots did not differ from that produced by multi-species plots (Table 3). Although values did not significantly differ, by early September fallow food plots produced 40% more invertebrate biomass than did multi-species plots. In fact, multi-species plots produced less ($P > 0.05$) biomass than did any other food plot type except native vegetation in early September (Table 3).

In the summer of 1999, there was week*food plot type interaction ($F_{66,327} = 1.41$, $P = 0.029$). Invertebrate biomass differed among weeks by food plot type ($F_{11,327} = 50.72$, $P < 0.001$) and among food plot types by week ($F_{6,327} = 48.23$, $P < 0.001$). Each food plot type significantly differed among weeks (fallow, $P = 0.026$; all other types, $P < 0.001$). Without exception, each food plot type produced much more invertebrate biomass in early summer than in late summer (Table 4; Figures 3a, b).

Similar to 1997, multi-species food plots produced more invertebrate biomass over the entire 1999 summer than did native vegetation or fallow plots (Table 4). However, different from 1997, invertebrate biomass differed among food plot types each

Table 2: Percent composition by weight of invertebrate orders collected in native vegetation (NaV) and food plots established by fallow disking only (FwD), fallow disking and fertilizing (FwF), or disking, fertilizing and planting a single species (i.e., browntop millet [BTM], iron and clay peas [ICP], or sorghum [SGM]), or a multi-species mix (i.e., browntop millet, catjang peas, iron and clay peas, Japanese millet, and pearl millet [MSP]) in the Pineywoods of east Texas during spring and summer, 1997 and 1999. The others category included Acarina, Anoplura, Coleoptera, Lepidoptera, Neuroptera, Odonata, Plecoptera, Psocoptera, Thysaboptera, and Trichoptera.

		Food plot type							
Year	Order	NaV	FwD	FwF	BtM	SGM	ICP	MSP	% composition $\bar{X}$
1997	Araneida	14	9	11	10	4	11	5	9.2
	Diptera	6	7	11	7	8	6	9	7.8
	Hemiptera	13	26	28	41	34	33	37	30.4
	Homoptera	25	23	23	23	33	23	26	25.2
	Hymenoptera	8	2	4	2	3	3	8	4.4
	Orthoptera	27	24	13	11	13	12	10	15.2
	Others/unknown	7	9	10	6	5	12	5	7.8
1999	Araneida	17	11	11	8	6	9	6	9.7
	Diptera	8	6	8	0	8	6	7	6.2
	Hemiptera	15	32	25	44	31	19	18	26.3
	Homoptera	25	26	26	30	34	47	46	33.4
	Hymenoptera	8	2	3	2	5	3	7	4.3
	Orthoptera	20	15	9	10	9	9	10	11.7
	Others/unknown	7	8	18	6	7	7	6	8.4

week except the last week of the 1999 study period. As with 1997, by mid-July, biomass values for fallow food plots were similar to values for food plots which had been cultivated (Table 4).

Invertebrate Richness

In 1997, the mean number of taxa per sample was 6.45 (range 4.6-7.4). There was no week*food plot interaction ($F_{30,159} = 1.55$, $P = 0.059$), but mean numbers of taxa did differ among food plot types ($F_{6,159} = 2.66$, $P = 0.017$) and among the 6 weekly samples ($F_{5,159} = 3.62$, $P = 0.004$). More taxa were recorded in samples from fallow plots ($\bar{x} = 6.9$) than from multi-species plots ($\bar{x} = 5.8$); means from the remaining food plot types overlapped both values. Taxon richness was lower the week of 9 July ($\bar{x} = 5.1$) than the weeks of 26 June, 23 July, 6 and 20 August, and 3 September (range 6.5-7.2).

The mean number of taxa per sample in 1999 was 6.55 (range 5.4-7.0). There was a week*food plot interaction ($F_{30,159} = 1.79$, $P = 0.012$). Taxon richness differed among food plot types ($F_{6,159} = 2.44$, $P = 0.028$) and among weeks ($F_{6,159} = 4.15$, $P = 0.001$). Taxon richness was higher in native vegetation ($\bar{x} = 6.8$) than in the fallow food plot types ($\bar{x} = 6.3$). As with 1997, values for the remaining food plot types overlapped both native vegetation and fallow values. Taxon richness in the week of 21 June ($\bar{x} = 6.1$) was lower than in the weeks of 6 and 20 July ($\bar{x} = 6.7$ for each) and 10 August ($\bar{x} = 6.8$), but similar to the weeks of 24 August ($\bar{x} = 6.5$) and 6 September ($\bar{x} = 6.5$).

Table 3: Mean biomass (g) of invertebrates collected in native vegetation and food plots established by fallow disking only, fallow disking and fertilizing, or disking, fertilizing, and planting a single species (i.e., browntop millet, iron and clay peas, or sorghum), or a multi-species mix (i.e., browntop millet, catjang peas, iron and clay peas, Japanese millet, and pearl millet) in the Pineywoods of east Texas during spring and summer, 1997. Within rows, means followed by the same letter did not differ P >0.05).

	Food plot type									
Week	Native vegetation	Fallow	Fallow with fertilizer	Browntop millet	Sorghum	Iron and clay peas	Multiple species	$\bar{x}$	F	P
26-Jun	0.2108b	0.2421b	0.3245ab	0.5352ab	0.5787ab	0.4810ab	0.7197a	0.4417	4.06	0.005
3-Jul	0.2377	0.2762	0.3213	0.4136	0.4983	0.3331	0.6372	0.3882	2.35	0.059
9-Jul	0.2070	0.2094	0.2753	0.3485	0.5528	0.3085	0.5788	0.3543	1.50	0.227
16-Jul	0.2314	0.3785	0.3918	0.4871	0.6529	0.4047	0.7151	0.4659	2.29	0.064
23-Jul	0.1730b	0.3092ab	0.3698ab	0.4531ab	0.5221ab	0.4345ab	0.5674a	0.4042	3.17	0.017
30-Jul	0.2167	0.3717	0.3425	0.3644	0.4318	0.5218	0.5177	0.3952	1.25	0.313
6-Aug	0.1884	0.3445	0.4284	0.4021	0.4363	0.3432	0.4160	0.3655	1.59	0.186
13-Aug	0.1710	0.5016	0.4163	0.3502	0.4110	0.3934	0.4516	0.3850	1.51	0.211
20-Aug	0.1310	0.5509	0.4631	0.4455	0.4121	0.3390	0.4981	0.4057	2.21	0.072
27-Aug	0.1193b	0.5671a	0.4784ab	0.4785ab	0.4504ab	0.5512a	0.5125a	0.4511	3.24	0.015
3-Sep	0.1132b	0.6038a	0.4490ab	0.5488ab	0.7234a	0.6259a	0.4319ab	0.4994	3.36	0.014
$\bar{x}$	0.1818c	0.3959b	0.3873b	0.4388ab	0.5154ab	0.4306ab	0.5496a			

Table 4: Mean biomass (g) of invertebrates collected in native vegetation and food plots established by fallow disking only, fallow disking and fertilizing, or disking, fertilizing, and planting a single species (i.e., browntop millet, iron and clay peas, or sorghum), or a multi-species mix (i.e., browntop millet, catjang peas, iron and clay peas, Japanese millet, and pearl millet) in the Pineywoods of east Texas during spring and summer, 1999. Within rows, means followed by the same letter did not differ ($P > 0.05$).

	Food plot type									
Week	Native vegetation	Fallow	Fallow with fertilizer	Browntop millet	Sorghum	Iron and clay peas	Multiple species	$\bar{x}$	F	P
21-Jun	0.0867c	0.2061bc	0.4792ab	0.6177a	0.5060ab	0.4834ab	0.6681a	0.4353	7.77	<0.001
28-Jun	0.1155b	0.4066ab	0.5339ab	0.6748a	0.3823ab	0.5242ab	0.5304ab	0.4525	3.39	0.012
6-Jul	0.1215c	0.3207bc	0.4902abc	0.6645ab	0.5326ab	0.5500ab	0.8556a	0.5050	6.06	<0.001
13-Jul	0.1051c	0.3482bc	0.4695ab	0.5248ab	0.5721ab	0.4886ab	0.8207a	0.4756	9.06	<0.001
20-Jul	0.0893b	0.2510a	0.4337a	0.4072a	0.3542a	0.4181a	0.4326a	0.3409	11.70	<0.001
4-Aug	0.0515b	0.4754a	0.3825a	0.5760a	0.3893a	0.4890a	0.6249a	0.4269	11.64	<0.001
10-Aug	0.0810b	0.2854a	0.3042a	0.2857a	0.3151a	0.3078a	0.3487a	0.2754	3.77	0.007
17-Aug	0.0328b	0.2394a	0.2186a	0.2882a	0.2502a	0.2798a	0.2756a	0.2264	6.55	<0.001
24-Aug	0.0289b	0.1864a	0.1232ab	0.1446ab	0.1540ab	0.1398ab	0.1843a	0.1373	2.91	0.025
31-Aug	0.0199b	0.0749ab	0.0943ab	0.1264a	0.1128ab	0.0835ab	0.1161a	0.0897	2.78	0.030
6-Sep	0.0179b	0.1502a	0.0467ab	0.1335a	0.0877ab	0.1185ab	0.0572ab	0.0874	3.35	0.018
14-Sep	0.0192	0.0676	0.0550	0.1507	0.0779	0.0851	0.0713	0.0753	2.08	0.088
$\bar{x}$	0.0641f	0.2500cde	0.3026acde	0.3828abc	0.3112acde	0.3307abcd	0.4155ab			

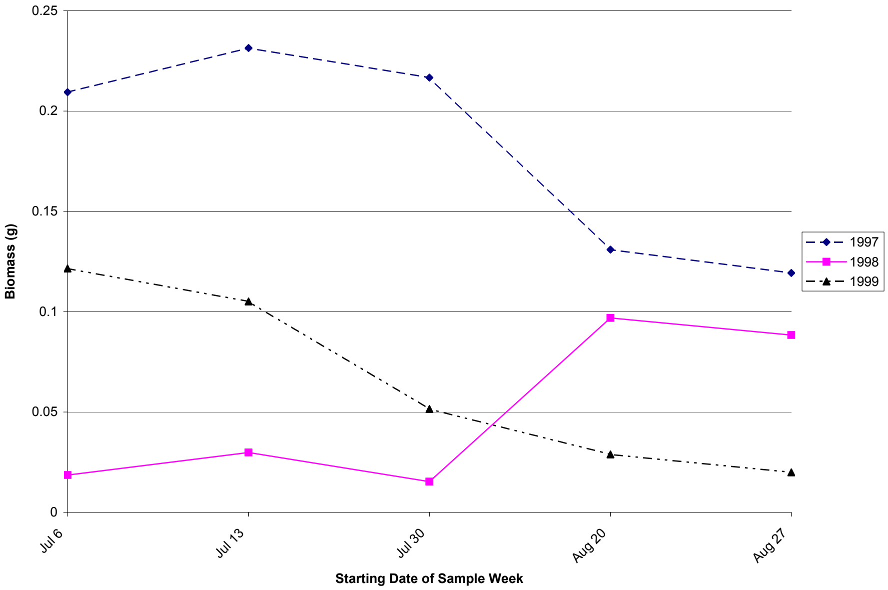

Figure 1: Invertebrate biomass (g) collected in native vegetation in the Pineywoods of east Texas during 5 weekly sampling periods in July and August, 1997, 1998, and 1999.

Discussion

Our results demonstrate the impacts of precipitation on invertebrate biomass in low-growing herbaceous vegetation. June, July, and August 1997 precipitation totals were below the 30-year average each month. However, rainfall was evenly distributed across the 3 months, and biomass remained relatively high throughout the summer. Conversely, the spring drought in 1998 resulted in virtually no vegetation, thus no invertebrates, in cultivated food plots during June and July. However, precipitation in late July and August resulted in relatively high invertebrate biomass in native vegetation by late August. The late July and August 1999 drought resulted in very low invertebrate biomass in late August and early September. These differences among summers are similar to the finding of Burger et al. (1993) in Missouri. They attributed different biomass values from 2 summers to different precipitation regimes. In retrospect, inexpensive rain gauges on each plot may have allowed us to better explain the relationship between the precipitation regime and invertebrate biomass.

When spring rains were sufficient, multi-species food plots consistently produced more invertebrate biomass than either fallow or native vegetation plots during late June and early July. However, by mid-July, vegetation in fallow plots had matured such that invertebrate biomass there equaled or exceeded that in multi-species plots. Although not statistically significant, similar patterns were detected between single-species and fallow plots. Generally, single-species plots had greater biomass than fallow plots in early summer but not late summer, and greater biomass than native vegetation throughout the summer.

With weeks pooled, all cultivated food plot types produced more invertebrate biomass than native vegetation in summer 1997 and 1999. Likewise, cultivated plots generally produced more invertebrate biomass than native vegetation each week. These results parallel those of Parsons et al. (2000*a*), who found greater invertebrate biomass in food plots than in native vegetation on the QMA.

During the summers of 1997 and 1999, invertebrate biomass was generally higher in multi-species food plots than in native vegetation; single-species plots showed similar trends. Likewise, summer-

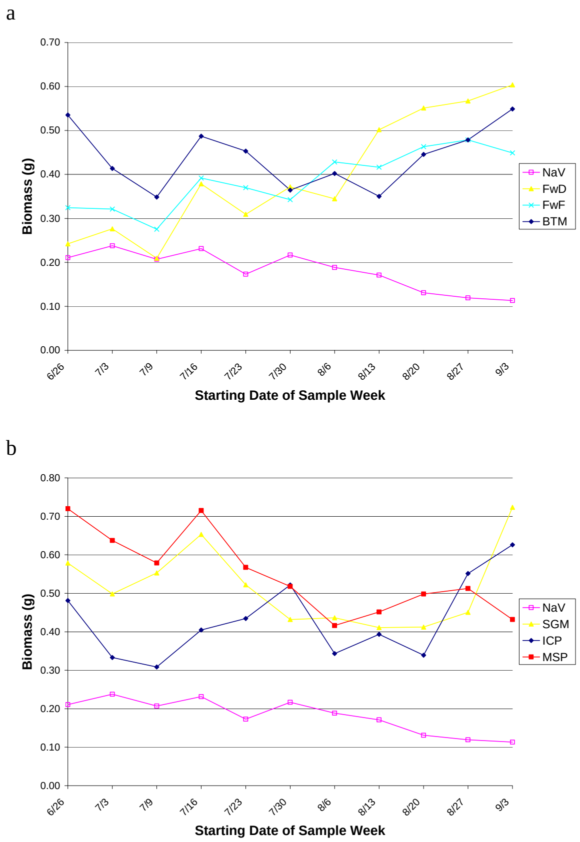

Figure 2: Invertebrate biomass (g) in native vegetation (NaV) and food plots established (a) by fallow disking only (FwD), fallow disking and fertilizing (FwF), or disking, fertilizing, and planting browntop millet (BTM); (b) by disking, fertilizing, and planting a single species (sorghum [SGM], iron and clay peas [ICP]), or a multi-species mix (browntop millet, catjang peas, iron and clay peas, Japanese millet, and pearl millet [MSP]) in the Pineywoods of east Texas during spring and summer,1997.

long biomass was slightly higher in multi-species plots than single-species plots. These differences may have been due to legumes in the multi-species plots. We did not quantify vegetation in food plots, but Burger et al. (1993) recorded more invertebrate biomass in CRP fields planted to red clover than in fields planted to 5 other non-legume species or commercial soybean fields. Conversely, in Mississippi, invertebrate biomass did not differ among Kobe lespedeza, old field, or fertilized old field plots (Jackson et al. 1987).

In early summer, we found few differences in biomass among plots that had been fertilized or fertilized and planted. However, biomass values in such plots were generally higher than biomass in fallow only (i.e., not fertilized) plots. These findings suggest that the application of fertilizer has a greater affect on invertebrate biomass in early summer than

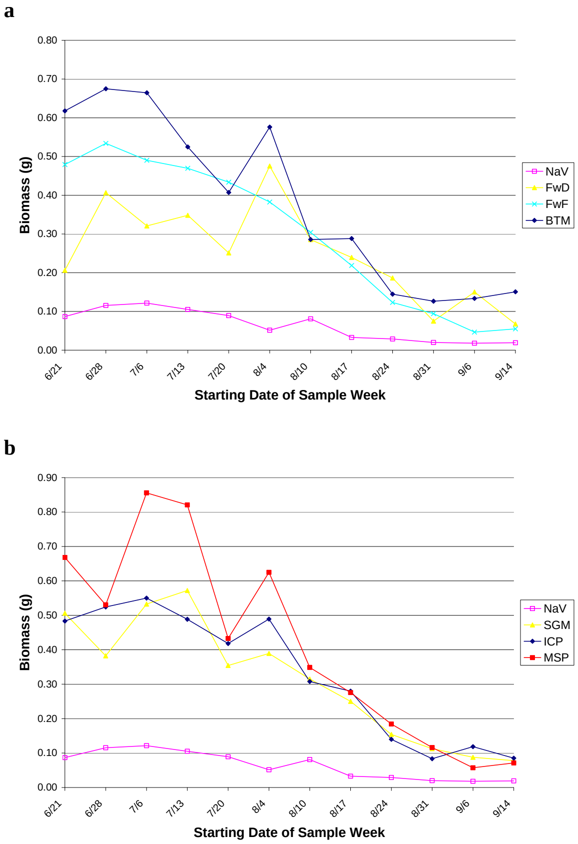

Figure 3: Invertebrate biomass (g) in native vegetation (NaV) and food plots established (a) by fallow disking only (FwD), fallow disking and fertilizing (FwF), disking, fertilizing, and planting browntop millet (BTM); (b) by disking, fertilizing, and planting a single species (sorghum [SGM] or iron and clay peas [ICP]), or a multi-species mix (browntop millet, catjang peas, iron and clay peas, Japanese millet, and pearl millet [MSP]) in the Pineywoods of east Texas during spring and summer,1999.

does the species planted in the food plots.

Although we compared numbers of taxa among food plot types by year, we made no attempt to compare biomass of each invertebrate taxum among food plot types either year. However, Hemiptera and Homoptera dominated biomass samples in cultivated plots and exceeded 50% in all fertilized plot types each year. In Mississippi, fertilized old field and Kobe lespedeza plots produced relatively high biomass of the same orders (Jackson et al. 1987), and in Georgia, millet and sorghum plots had relatively high biomass of Hemiptera, Homoptera, and Hymenoptera (Maidens and Carroll 2002). During a 2-year study in Missouri, Hemiptera and Homoptera biomass values were much higher in red clover fields than in other CRP or soybean fields each year. Within red clover fields, Homoptera made up the highest proportions of biomass each year, while Hemiptera or Orthoptera ranked second (Burger et al. 1993).

Management Implications

Land owners and managers establish food plots for many purposes (e.g., food for other game species, erosion control, road stabilization). As food plots are seldom established solely to benefit bobwhite chicks, our finding and recommendations should be modified to meet other objectives. Regardless of how plots are established or what is planted in them, the precipitation regime is critical and unpredictable. When spring precipitation is adequate, multi-species food plots provide the greatest invertebrate biomass during the peak hatching period for bobwhite chicks in June and early July. At that time of summer, the mixture of plant species may also provide better overhead protection for hens and chicks than other food plot types. Also, the multi-species plots are more likely to meet other objectives of landowners than are single-species plots. If other factors (e.g., cost, time constraints) are a consideration, single-species and fallow disking with fertilizing food plots produce only slightly less invertebrate biomass than multi-species plots.

During a year of average rainfall, fallow disked food plot types have as much invertebrate biomass as planted and/or fertilized types of plots by mid- to late summer; in dry summers, biomass in fallow disked plots may exceed that in other types of food plots. However, native plant species in fallow plots may provide less overhead protection than is provided by multi-species plots. Regardless, a combination of multi-species (with legumes) food plots and fallow disking should provide invertebrates for bobwhite chicks throughout the summer.

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Integrating Burning and Insecticide to Reduce Fire Ant Impacts on Bobwhite Chicks

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Hatching northern bobwhite (*Colinus virginianus*) chicks experience a low survival rate when exposed to a significant number of foraging red imported fire ants (*Solenopsis invicta*; RIFA). We initiated a study in southeastern Texas to determine if a reduced rate of insecticide and/or prescribed burning could decrease the foraging activity of RIFA below the threshold that causes mortality of northern bobwhite chicks. Research sites were divided into burned and nonburned plots and individual plots randomly received one of 4 rates of insecticide treatment: 0, 50, 75 or 100% of the recommended label rate (1.68 kg/ha) of Amdro® (hydramethylnon) insecticide bait (Ambrands, Atlanta, GA). Bait cup sampling of RIFA was conducted and differences in RIFA foraging activity were analyzed among treatments. As the rate of Amdro® application increased, RIFA foraging activity declined. Data from 2002 and 2003 revealed a difference in mean number of foraging RIFA in insecticide treated plots versus control plots ($P < 0.05$) when testing for the main effect of insecticide treatment. The mean number of foraging RIFA in 2002 decreased approximately 34%, whereas the mean number of foraging RIFA in 2003 decreased approximately 39%. In both years, the mean number of foraging RIFA collected in bait cups in burned plots was not different from nonburned plots ($P > 0.05$).

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Key words: Amdro®,*Colinus virginianus*, , northern bobwhite, prescribed fire,red imported fire ant, *Solenopsis invicta*, Texas

Introduction

The red imported fire ant (*Solenopsis invicta* Buren; Hymenoptera: Formicidae) is an exotic species of particular concern to the southern United States. Extensive observational and experimental evidence has chronicled and quantified the expansion and subsequent environmental damage caused by *S. invicta* (Lofgren 1986). Ground nesting species such as northern bobwhite have received much attention due to the potential negative impacts from RIFA. Allen et al. (2000) reported a positive correlation between the years of RIFA infestation and bobwhite population declines in certain Texas counties. In addition, Mueller et al. (1999) recorded a lower survival rate among hatching northern bobwhite when exposed to RIFA assaults. Protecting the hatching bobwhite chicks from RIFA increased survival rate to 21 days of age. Furthermore, chick survival was directly related to the quantity (n = 300) of foraging RIFA captured within a 30-minute period in a standardized bait cup placed in the nests of northern bobwhites the day after hatch (Mueller et al. 1999). Controlling RIFA using insecticides appears to be a simple fix for the problem, but control methods for *S. invicta* have their limitations.

Methods for managing RIFA are expensive, labor intensive, and provide only temporary RIFA reductions. Numerous commercial insecticides are available for controlling RIFA such as Amdro® and Logic® insecticide baits (Collins et al. 1992). Chemical insecticides such as these offer some level of control, but are not economically feasible for most landowners to apply to large areas. Consequently, natural control methods for slowing RIFA activity are currently being investigated. Environmental dis-

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turbances such as prescribed burning may present a form of RIFA control. Forbes et al. (2002) observed a potential negative impact of prescribed burning on RIFA mound density. He hypothesized this negative impact was caused by reduced soil moisture and food availability that follows fire until regrowth occurs. Forbes et al. (2002) theorized that the negative consequences of burning on RIFA colonies were merely short term.

Prescribed burning is a relatively low cost management tool used to improve wildlife habitat. Implementing prescribed burning can specifically benefit northern bobwhite by encouraging forb production and controlling vegetation density in productive areas (Buckner and Landers 1979, Hansmire et al. 1988). Wright and Bailey (1982) suggested that protein-rich insects and seeds are plentiful in burned areas, providing important food resources for northern bobwhites. Prescribed burning could be utilized to increase the effectiveness of insecticide to negatively impact the foraging activity of RIFA. Literature quantifying RIFA responses to soil moisture and temperature following prescribed burning is limited. A thorough understanding of the relationships among these factors and prescribed burning would provide important information concerning the timing of the integration of burning and insecticide treatments to reduce RIFA foraging activity.

Managing, rather than eradicating RIFA is a strategy that can be used to reduce RIFA impacts on northern bobwhite chicks (Mueller et al. 1999). Mueller et al. (1999) reported a "threshold level" of foraging ants related to northern bobwhite chick survival. He reported that chick survival rates approached zero when chicks were exposed to 300 or more foraging RIFA. When the numbers of foraging RIFA fell below 300 ants per bait cup, northern bobwhite chicks were not threatened. This information is valuable because northern bobwhites may co-exist in RIFA-infested areas, but RIFA management is needed to improve chick survival. Our objective was to evaluate a management strategy using prescribed burning and/or a reduced amount of insecticide to decrease the foraging activity of red imported fire ants below the threshold level that causes mortality of northern bobwhite chicks. Chemical methods for managing RIFA can be costly for landowners, especially when applied to large areas. An integrated method combining a reduced amount of insecticide with prescribed burning addresses this problem of cost-effective RIFA control. Data from Forbes et al. (2002) and Mueller et al. (1999) prompted this study to examine integrated RIFA management using prescribed burning in conjunction with insecticide application. The goal of our research is to provide an economically viable management strategy to landowners for limiting RIFA impacts on northern bobwhite chicks.

Study Area

Research sites for this project were established in early spring 2002 and 2003 in actively cattle-grazed pastures. Study sites were selected based on the following criteria: adequate fine fuel load and continuous fine fuel to carry prescribed fires uniformly across research plots. Additionally, based on preliminary ant sampling, study sites displayed evidence of sufficient RIFA densities to recruit more than 300 RIFA using the standardized bait cup method (Mueller et al. 1999).

Research was conducted in Goliad and Victoria Counties, Texas in 2002 and Calhoun County, Texas in 2003. These counties are part of the Texas Coastal Prairie and consist of clay, clay loam, loam, and sandy loam soils with level to gently sloping landscapes. Dominant vegetation includes little bluestem (*Schizachyrium scoparim*), huisache (*Acacia smallii*), mesquite (*Prosopis glandulosa*), and prickly pear (*Opuntia lindheimeri*). The climate in these areas is humid and subtropical with mild winters. Average daily summer temperature is 28.5°C. Annual precipitation ranges between 78.74 and 113.18 cm (Miller 1982, Mowry and Bower 1978).

Methods

Research Plot Design

We established 8, 150 x 300-m blocks in spring 2002 in Victoria and Goliad Counties using a ran-

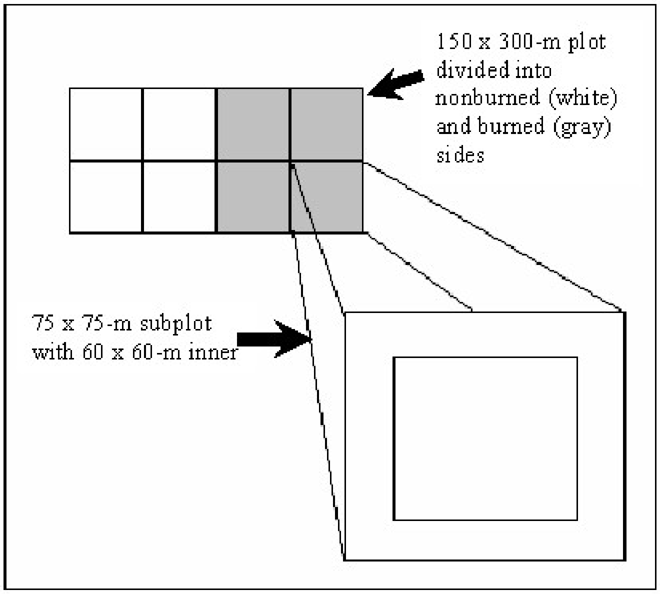

Figure 1: Research plot design.

domized block design. Each 150 x 300-m block consisted of 2, 150 x 150-m paired plots (1). One, 150 x 150-m plot within each block was randomly chosen for prescribed burning treatment. Subsequent to burning, each plot was stratified to create 4, 75 x 75-m subplots (8 subplots per block). We randomly selected each subplot to receive one of 4 rates of insecticide treatment: 0, 50, 75 or 100% of the recommended label rate (1.68 kg/ha) of Amdro® (hydramethylnon) insecticide bait (Ambrands, Atlanta, GA). For the purpose of this study, we define a "control" plot as receiving 0% of the label rate of Amdro®. We applied insecticide to a 60 x 60-m core area within each subplot to reduce impacts to adjacent treatment areas. Plot dimensions were developed based on the foraging distance of a fire ant colony. Subplot size was large enough to prevent invasion of the core area by ant colonies from outside the subplot (Mueller et al. 1999). Treatments were repeated in 2003 on 6 previously untreated plots in Calhoun County. We established only 6 research plots, or replicates, in 2003 due to an insufficient amount of burned area available.

Treatment Application

Prescribed burn treatments were randomly assigned to plots. Prescribed burn treatments were completed between January and March at all research sites during both years. We applied insecticide treatments following initial RIFA sampling. Application occurred during 10-15 May 2002 and 17-21 May 2003. We timed insecticide application to coincide ant control with peak bobwhite hatching season in southern Texas (Lehmann 1984). Amdro® was applied using a Herd GT-77 broadcast spreader (Herd Seeder Company, Inc., Logansport, IN) mounted to the rear of a Yamaha ATV (Yamaha Motor Corporation, U.S.A.). In 2002, we controlled differences in the rate of insecticide application by varying ATV speed. These applications were calibrated by measuring the effective swath width of the spreader and the weight of insecticide delivered over a 50 m distance at different speeds. This insecticide application method was modified in 2003 to reduce the risk of driving the ATV at high speeds across uneven terrain. While the application method was modified during 2003, the individual application rates remained the same. In 2003, to insure

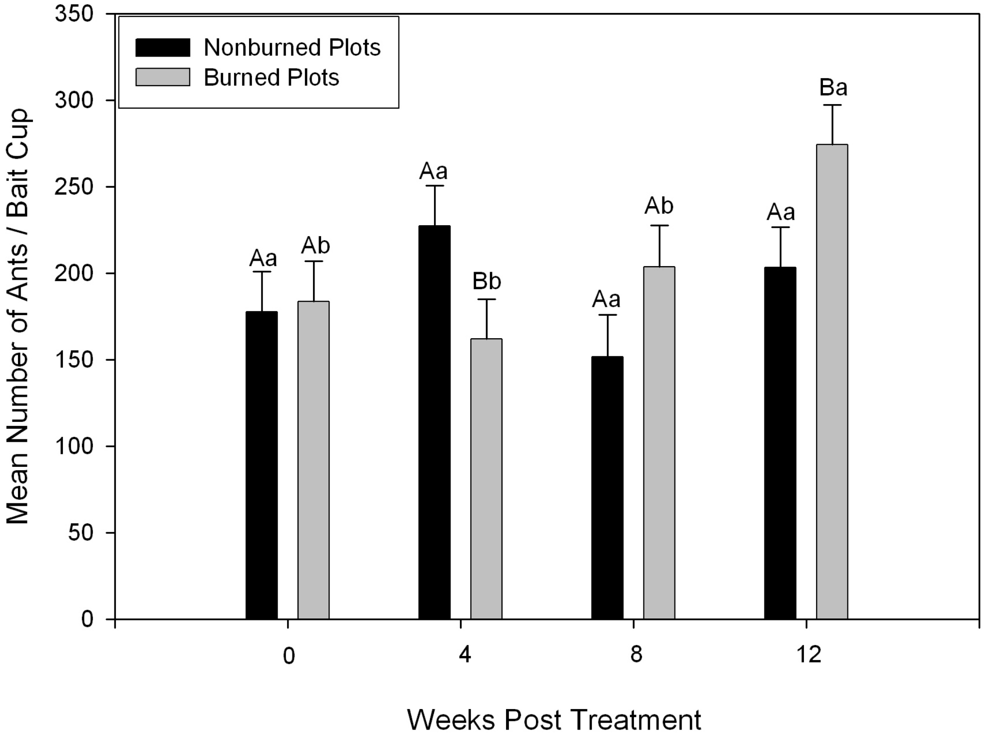

Figure 2: Mean (±SE) number of foraging RIFA collected in bait cups from burned and nonburned plots in Goliad and Victoria Counties, Texas, in 2002. Means within a sampling time followed by the same upper case letter are not significantly different ($P > 0.05$). Means within a burning treatment followed by the same lower case letter are not significantly different ($P > 0.05$).

uniform distribution across the entirety of each subplot, we mixed each individual rate of insecticide with Quaker® corn grits (Quaker Oats Company, Chicago) to provide a total mixture (Amdro® and Quaker® corn grits) weighing 0.91 kg. This mixture weight is the amount delivered when traversing the entire core area of the subplot (at the effective swath width) 1 time at 9.7 km/h. Since corn grits comprise approximately 75% g/g of Amdro® insecticide bait, the additional corn grit effectively diluted the concentration of insecticide without fundamentally changing the bait. This allowed for the reduced speed of the ATV while still achieving the same insecticide application rate.

Ant Sampling

Efficacy of treatments on RIFA foraging activity within the core area of each subplot was evaluated using a bait cup method as previously described (Porter and Tschinkel 1987, Mueller et al. 1999). A hotdog bait (approximately 5-mm slice) was placed inside a 28.4 ml plastic cup and then positioned on bare ground. We took a random number of steps out into each subplot core area and placed 5 bait cups approximately 10 meters apart along a diagonal transect. After 30 minutes, the plastic bait cup, including all ants contained inside, was collected and placed in a specimen container. Ant samples were placed in alcohol until specimens could be identified and counted. Bait cup sampling was conducted immediately prior to insecticide treatment, 4 weeks, 8 weeks, and 12 weeks after treatment. In 2002, ant sampling occurred during 10-15 May, 10-13 June, 8-11 July, and 5-8 August. In 2003, ant sampling occurred during 17-21 May, 16-19 June, and 11-

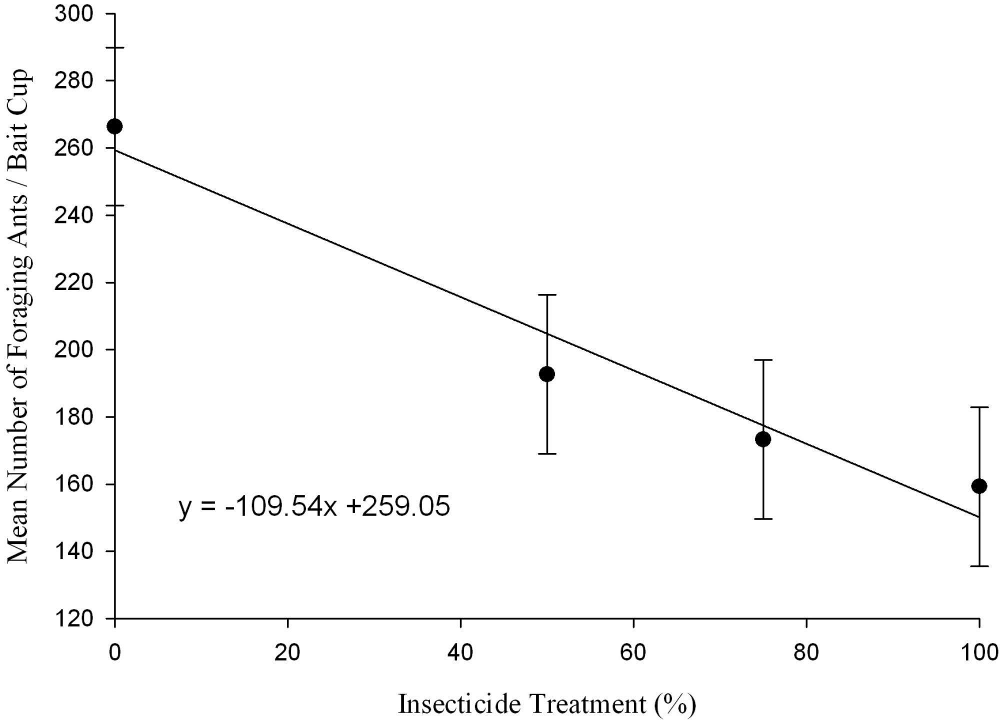

Figure 3: Mean (±SE) number of foraging RIFA collected in bait cups plotted against percent label rate of Amdro®insecticide treatment. Ants were collected from plots in Goliad and Victoria Counties, Texas, in 2002.

14 August. We could not sample ants 8 weeks post-treatment in 2003 due to Hurricane Claudette; therefore, there is no data for that sampling time. Ant sampling occurred between 0800 hours and 1200 hours, with air temperatures ranging between 21°C and 32°C, and soil temperatures ranging between 22°C and 31.67°C.

Data Analysis

We used a randomized block design to analyze the data. The variable of interest was mean number of foraging RIFA collected in bait cups. Differences in RIFA foraging activity among treatments were assessed a using two-factor repeated measure ANOVA with burning and insecticide as the two main factors. Differences among individual means were determined using the least squares method. The relationship between insecticide application rate and mean number of foraging RIFA collected in bait cups was evaluated using linear regression.

Results

2002 Results

In 2002, the mean number of foraging RIFA collected in bait cups in burned plots did not differ from unburned plots (ANOVA, F = 1.14, df = 1, P = 0.2864). In addition, any differences observed among the simple main effects were not consistent (2). For example, RIFA means 0 and 8 weeks post-treatment were not different between burned and nonburned plots. However, RIFA means were lower in burned plots as compared to nonburned plots 4 weeks post-treatment, but higher 12 weeks post-treatment. Furthermore, no differences were detected among nonburned plot means throughout the sampling period in 2002 (2). The mean number of foraging RIFA collected in bait cups was different among sampling times (ANOVA, F = 3.64, df = 3, P

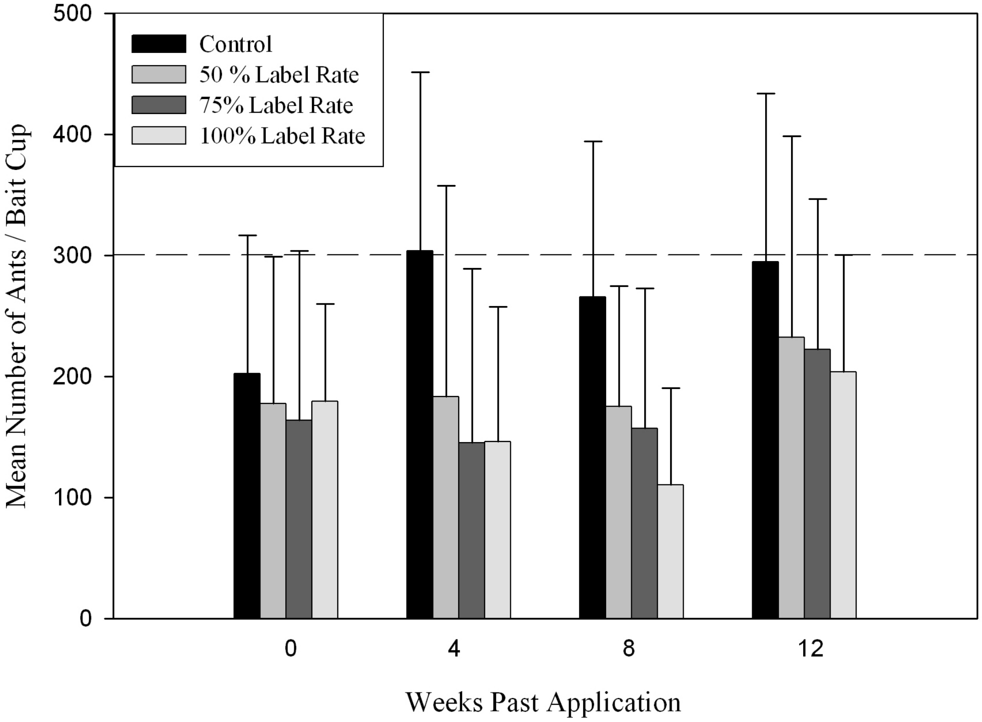

Figure 4: Mean (±SE) number of foraging RIFA collected in bait cups for each insecticide treatment within four different sampling periods. Dashed line indicates threshold level of 300 foraging ants. Ants collected from plots in Goliad and Victoria Counties, Texas, in 2002.

= 0.0136). However, an interaction occurred between burning treatments and sampling times (ANOVA, F = 4.29, df = 3, P = 0.0058); therefore, variations over time were different in burned and nonburned plots.

Insecticide treatment application rate was negatively related to the mean number of foraging RIFA (F = 30.06, df = 1, $P < 0.0001$; 3). The mean number of foraging RIFA collected in bait cups was different in insecticide-treated plots versus control plots (zero percent insecticide application rate) (ANOVA, F = 10.49, df = 3, $P < 0.0001$) when testing for the main effect of insecticide treatment. However, individual insecticide application rate means were not different from each other (4). Each standard deviation bar for the control means exceeded 300 foraging ants (threshold level for bobwhite chick survival). Furthermore, each insecticide mean fell below this threshold level of 300 foraging ants.

2003 Results

The results from 2003 were similar to 2002. In 2003 the mean number of foraging RIFA collected in bait cups in burned plots did not differ from unburned plots (ANOVA, F = 0.13, df = 1, P = 0.7387; 5). Fire ant foraging means differed among sampling times (ANOVA, F = 77.51, df = 2, $P < 0.0001$). The mean number of foraging RIFA decreased as time increased. As in 2002, insecticide treatment application rate was negatively related to the mean number of foraging RIFA (F = 7.53, df = 1, $P < 0.0105$; 6). The mean number of foraging RIFA collected in bait cups was different in insecticide-treated plots versus control plots (ANOVA, F = 2.88, df = 3, P = 0.05) when testing for the main effect of insecticide treatment. However, means of individual insecticide application rates were not different from each other (7). Each standard deviation bar for the control means

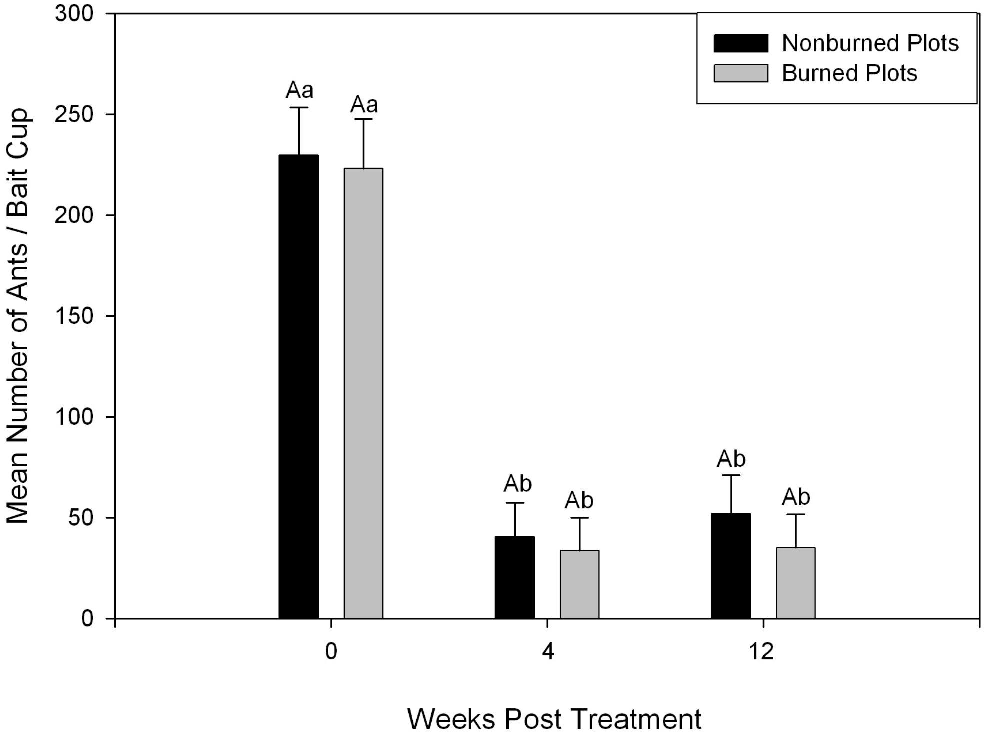

Figure 5: Mean (±SE) number of foraging RIFA collected in bait cups from burned and nonburned plots in Calhoun County, Texas, in 2003. Means within a sampling time followed by the same upper case letter are not significantly different ($P > 0.05$). Means within a burning treatment followed by the same lower case letter are not significantly different ($P > 0.05$).

exceeds 300 foraging ants (threshold level for bobwhite chick survival). Furthermore, each insecticide mean fell below this threshold level of 300 foraging ants.

Discussion

Red imported fire ant foraging activity was reduced by Amdro® insecticide treatments. As the rate of Amdro® increased, foraging activity of RIFA declined. Our results are similar to Apperson et al. (1984) who reported that Amdro® was effective in decreasing the number of foraging RIFA workers. In 2003, RIFA foraging means decreased considerably between 0 weeks post-treatment and 4 weeks post-treatment. This change is much greater than what was observed during these same time periods in 2002. Variation in foraging activity is most likely due to environmental conditions during insecticide application. Lack of foraging activity can be attributed to reduced soil moisture or unsuitable soil temperatures (Porter and Tschinkel 1987). Foraging means in 2003 are also much lower in 12 weeks post-treatment compared to 2002 data. Hurricane Claudette might explain this shift in foraging activity. Hurricane Claudette hit the Texas coast in July 2003 (8 weeks post-treatment). This extreme weather event could have negatively impacted RIFA populations by reducing their foraging activity and causing dispersal from low lying areas (Rhoades and Davis 1967).

In our study, Amdro® was successful in decreasing RIFA foraging activity below the threshold level of 300 ants per bait cup (Mueller et al. 1999). Controlling RIFA activity below this threshold level has

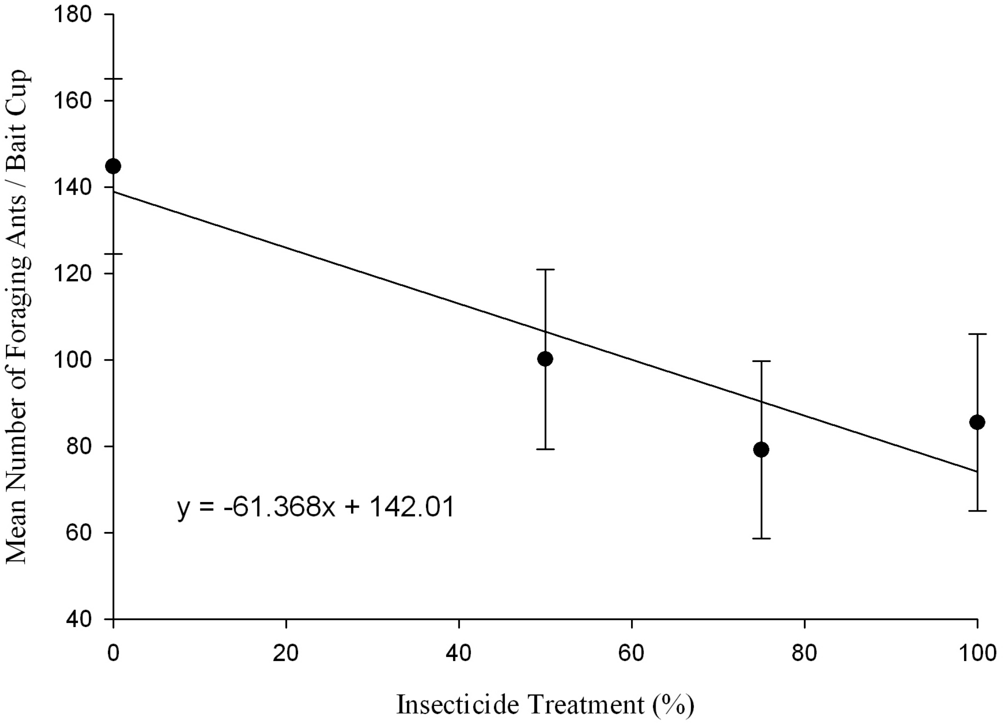

Figure 6: Mean (±SE) number of foraging RIFA collected in bait cups plotted against percent label rate of Amdro®insecticide treatment. Ants were collected from plots in Calhoun County, Texas, in 2003.

been positively related to bobwhite quail chick survival (Mueller et al. 1999). It is especially important to note that all three rates of Amdro (50%, 75%, and 100%) were equally effective in achieving this objective. Our study produced similar results as earlier research that examined the effectiveness of reduced rates of insecticides. Drees et al. (1993) found that a reduced rate of Logic® insecticide is as effective in suppressing RIFA as the full rate.

Our data indicate that a reduced rate of Amdro® insecticide is as effective as using a full label rate for reducing RIFA foraging activity and their impacts on northern bobwhite chicks in a Texas coastal prairie environment. This outcome is important to landowners who want to manage their land for northern bobwhite by controlling RIFA activity. Using a reduced rate of insecticide to decrease RIFA activity equates to lower management costs. For example, a landowner who uses 50% of the recommended label rate of Amdro as opposed to the full rate is able to treat twice as much land at the same cost. A 25-pound bag of Amdro insecticide has a retail cost of $250 (2003, personal observation in Lubbock, TX). Treatment cost, excluding labor and equipment, is approximately $37.00 per hectare when using 100% of the label rate of Amdro (1.68 kg/hectare). In comparison, when using 50% of the label rate of Amdro it costs approximately $18.50 per hectare to treat for RIFA.

Burning combined with insecticide application provided no additional benefit to insecticide application alone for decreasing RIFA foraging activity. Therefore, when landowners are attempting to control RIFA activity to benefit bobwhite chick survival, our data suggest there is no additional benefit to use prescribed burning. We initially theorized that using fire combined with insecticide application would significantly reduce RIFA foraging activity.

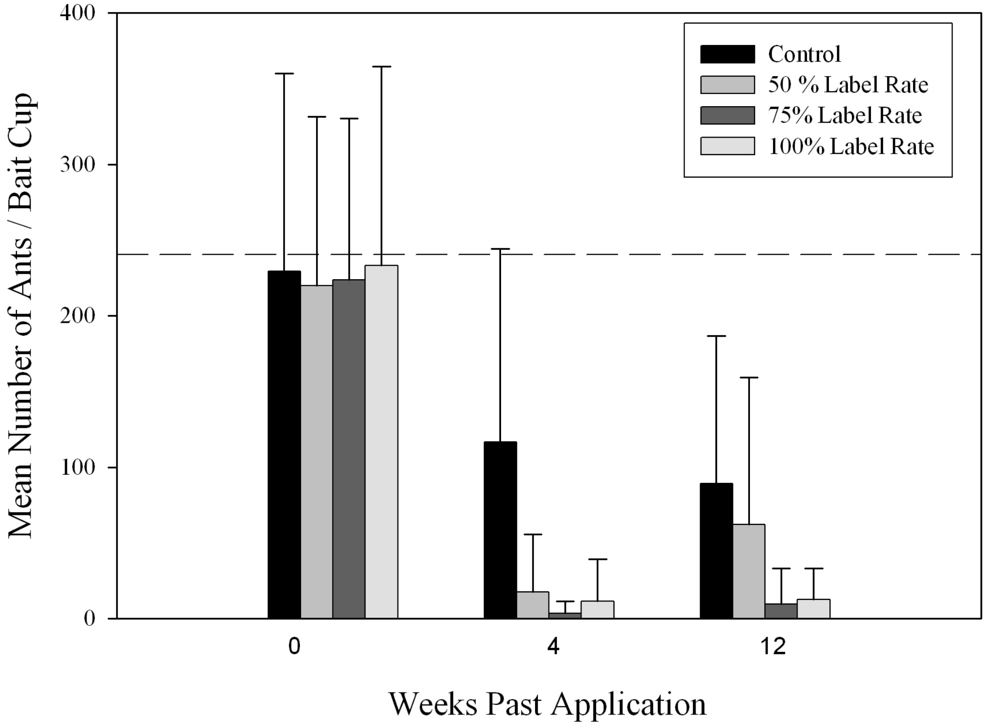

Figure 7: Mean (±SE) number of foraging RIFA collected in bait cups for each insecticide treatment within 3 different sampling periods. Dashed line indicates threshold level of 300 foraging ants. Missing data in 8 weeks post treatment due to hurricane. Ants collected from plots in Calhoun County, Texas, in 2003.

Although we found no decrease in RIFA following a prescribed burn treatment, we also did not see any increase in ant numbers.

No differences in burning versus nonburned treatment means could be explained by a number of factors. Forbes et al. (2002) hypothesized that prescribed burning negatively impacts RIFA because of reduced soil moisture and food availability that follows fire, until regrowth occurs. It is possible that the burns conducted for this study did not burn hot enough to make substantial impacts on forage availability or soil moisture. Another reason might be attributed to the RIFA recolonization success. Red imported fire ant colonies affected by these fires might have had adequate time to recover before the impacts of the insecticide took affect. Limited research is available on the effects of fire on *S. invicta*. Additional research is needed to better understand the relationship between habitat disturbances such as fire and RIFA activity.

Red imported fire ants can cause serious problems for wildlife such as northern bobwhite. Chicks are especially sensitive to RIFA activity and can benefit from fire ant control (Mueller et al. 1999). Eradication is not required to reduce RIFA impacts on bobwhite chicks, although RIFA control on some level is important (Mueller et al. 1999). Control methods need to be effective yet affordable for landowners. One of our objectives was to evaluate a more economical way for landowners to control RIFA impacts on northern bobwhite chicks. Our results suggest that using a reduced rate of Amdro® insecticide is effective in reducing RIFA foraging activity. Using a reduced rate of insecticide has a cost benefit for landowners treating for RIFA, but additional economic analysis of RIFA control is needed.

Eradication of RIFA may be an unrealistic goal; therefore, it is important to continue research to develop effective management strategies that will reduce RIFA impacts on important wildlife species such as the northern bobwhite.

Acknowledgments

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Northern Bobwhite Brood Habitat Selection in South Florida

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During the past 3 decades, Northern Bobwhite (*Colinus virginianus*; hereafter, bobwhite(s)) populations have decreased throughout most of their distribution. A variety of factors have been attributed as the cause for this decline including changes in land use, agricultural intensification, increased predation, and high chick mortality. We assessed fourth-order habitat selection of broods in south Florida to develop predictions of management strategies that favor bobwhite brood success. We analyzed canopy coverage at actual brood locations versus both random-within MCP home range locations and random-outside MCP home range locations. Average home range size was 5.53 ± 2.43 ha. Our data suggests that no single vegetation type can be used to predict use by bobwhite broods. The models we evaluated using Akaike's Information Criterion (AIC) supports this belief. We also observed sod-forming grasses and forbs as the most prevalent vegetation types at brood and random-within MCP home range locations. Broad-leaved woody vegetation and legumes were more prevalent at brood locations than random locations. Our research demonstrates that plant community diversity is likely more important than a single functional group of plants. We believe that, at the ranch level, a combination of vegetation management within pastures, as well as large-scale management increasing interspersion of desirable vegetation communities will provide bobwhites quality habitat during all periods of their life cycle.

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Key words: brood, *Colinus virginianus*, Florida, habitat selection, northern bobwhite, quail

Introduction

During the past 3 decades Northern Bobwhite (*Colinus virginianus*; hereafter, bobwhite[s]) populations have markedly decreased throughout most of their distribution (Droege and Sauer 1990, Brennan 1991, Church et al. 1993). In the southeastern U.S.A., bobwhite populations declined by 66% during 1966 to 1999 (Sauer et al. 2000). This decline has been attributed to changes in land-use associated with reforestation, suburban and urban sprawl, and agricultural intensification (Brennan 1991, Roseberry 1993). Still other reasons have been proposed for these declining trends including increases in avian and mammalian predators ((Rollins and Carroll 2001), introduction of the red-imported fire ants (RIFA; *Solenopsis invicta*), and increased use of pesticides among agricultural ecosystems (Palmer et al. 1998).

It is crucial to provide habitat that induces the recruitment of offspring into the population if that population experiences high rates of annual mortality such as bobwhites (Yates et al. 1995). Bobwhites experience high annual mortality in the Southeast; mortality rates range from 70%-80% (Speake 1967, Simpson 1976). Therefore, adequate chick survival is critical to the sustain bobwhite populations. The use of certain habitats does not necessarily mean higher survival in those habitats, but assumptions can be made about the importance of those habitats to bobwhites. It is important for landowners and wildlife biologists to know and understand more about brood habitat throughout the bobwhite's range so populations can be better managed for both conservation and recreation.

Micro-habitat selection of broods is the least-studied component of bobwhite ecology, and south Florida is perhaps the least-studied area of bobwhite range. Determining the most valuable habitat

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Table 1: Sum and average patch size of various cover types on the 2x4 Ranch near Arcadia, Florida, USA, 2005.

Cover type	Sum (ha)	Mean patch size (ha)
Fallow	177.72	4.44
Improved Pasture	1456.32	34.67
Other	35.09	3.9
Semi-improved Pasture	187.56	46.89
Wet Area	163.55	1.84
Wood	263.62	8.5
Young Grove	87.62	43.81

for bobwhite broods on this landscape will possibly lead to better management and higher population densities.

The objective of this study is to examine fourth-order habitat selection by bobwhites in southern Florida. Johnson (1980) describes a hierarchical nature of habitat selection: first-order selection is the geographical range of a species; second-order selection is the home range of an individual or social group; third-order selection is the use of habitat components within the home range; and fourth-order selection is micro-site plant species cover and composition selected from those available at the location. We predicted that broods would utilize habitats with more bunchgrass, forbs, and legumes and habitats with little to no sod-forming pasture grasses.

Study Area

This study was conducted on the 2x4 Ranch, which is located southeast of the peninsular town of Arcadia in Desoto County, Florida, U.S.A. The ranch supports a cattle operation with approximately 1,000 head of brood Brangus cows. The cattle are managed under an intensive rotationally grazed system. The ranch is dominated by improved pasture with the remaining portions being a mix of fallow, woody, wetland, and citrus groves (Table 1). Bahia grass (*Paspalum notatum*) is the dominant vegetation type throughout the improved pastureland (Table 1). The topography of the landscape is predominantly flat with a maximum change in relief of 3 meters. The presence of surface water continually changes depending on the day and season and is often altered mechanically by the use of irrigation ditches. Annual rainfall averages 135 cm. Since acquiring the land in the early 1980s, the landowners have anecdotally reported drastic declines in bobwhite populations.

Methods

During February 2005 to April 2005, we captured bobwhites using standard wire walk-in funnel traps baited with grain sorghum (Stoddard 1931). We banded and fitted birds with a 6.4-6.9 g pendant-style radio transmitter and released them. We located radio-marked bobwhites using homing techniques (White and Garrott 1990) about 5 days per week and approached them to within 10-25 m. When we found a bird in the same location two days in a row, we assumed it to be nesting. We marked and monitored the nest daily. We verified the presence of the nest and recorded the number of eggs when the bird was absent from the nest. After hatching, we monitored the brood daily. At 14 days, we flushed the brood to assure the adult bird was still attending the chicks. Only broods with verified chicks at 14 days were included in the study-chicks

Table 2: Description of variables measured for Northern bobwhite broods in Florida, USA 2005.

Variable	Description
BLACK	Blackberry; *Rubus* spp.
SOFG	Sod-forming grasses; Bahia grass (*Paspalum notatum*), Bermuda grass (*Cynodon dactylon*)
FORB	Forbs; Queen Anne Delight (*Stillingia sylvatica*), Dogfennel (*Eupatorium capillifolium*)
BUGR	Bunchgrasses; Wire grass (*Aristada stricta*), Broomsedge (*Andropogon virginicus*)
BLWD	Broad-leaf woody; Wax myrtle (*Myrica cerifera*)
LEGM	Legumes: Partridgepea, Desmodium, Sesbania (*Chameacrista*, *Desmodium*, *Sesbania* spp.)
LITT	Litter
BARE	Bare ground

at this age can fly fairly well and subsequent survival is perceived to be much higher.

We used the 100% minimum convex polygon extension in ArcView® 3.2 to map home ranges of each brood (Mohr 1947). Each brood had a minimum of 14 locations used in creating the home range. We assigned 10 random points within each home range and 10 random points outside of each home range for each of the five successful broods in this study. We measured the vegetation at both the observed locations and the random points using canopy coverage. We placed a 1-m^2 quadrat on the ground at the center of each point. We estimated percent canopy coverage for each of the following classes: bare ground, blackberry, bunch grass, broad-leaf woody, forb, legume, litter, and sod-forming grass (Table 2). *A priori* we believed blackberry to be an important component of brood habitat because of the cover/food resources it provides. We placed the quadrat at each of the four cardinal directions 3 m from the center point to account for potential telemetry error for the location. The mean percentages for each of the 5 quadrats were used to represent the vegetative characteristics for each location.

Data Analysis

Prior to analysis, we transformed data using arc-sin transformations to normalize the percentage data. However, descriptive statistics are reported, untransformed. Prior to modeling, we used a Pearson Correlation test statistic for each pair of predictor variables. Variables that were deemed correlated if $r^2 > 0.30$, thus, were eliminated from the analysis to avoid multicollinearity because multicollinearity creates unnecessary redundancy and over-fitting in models.

We used forward stepwise logistic regression ($P < 0.05$) to assess bobwhite brood habitat selection to random points at two spatial scales, within 100% MCP home range and outside home range to address two orders of habitat selection (third and fourth orders; Johnson 1980). The 100% MCP technique was used because of our limited sample size. This technique does not eliminate any locations from the sample. We conducted all regression analyses using PROC LOGISTIC (SAS Institute, Inc. 2003). We set the significance level at $P \leq 0.05$ within the stepwise procedure.

We used logistic regression analysis (Weisberg 1985) under a model selection (AIC) framework (Burnham and Anderson 2002). We developed *a priori* habitat models based on our experience and the ecology and biology of bobwhites. Our models only contained variables found to be significant in the stepwise procedure.

We used an information theoretic-approach ((Burnham and Anderson 2002), to evaluate how plausible the logistic regression models were at explaining brood habitat use. A global model was

Table 3: Mean ground cover (%) and 95% confidence interval of observed locations, random within MCP home-range, and random outside MCP home-range for brood rearing northern bobwhites in Florida, 2005. See Table 1 for variable descriptions.

	Brood Location		RandIn		RandOut	
Variable	Mean	CI	Mean	CI	Mean	CI
BLACK	2.64	9.52	1.16	9.02	0.78	12.59
SOFG	47.12	8.08	33.47	4.97	18.62	3.28
FORB	27.70	3.85	19.30	1.06	26.70	0.76
BUGR	17.56	5.29	11.08	5.24	21.40	4.53
BLWD	11.26	6.80	5.90	5.58	4.16	6.10
LEGM	4.66	1.96	1.14	0.54	0.58	0.39
LITT	5.66	2.70	8.98	2.67	6.38	2.31
BARE	8.90	3.47	8.53	2.85	5.64	3.11

constructed that included all non-correlated variables. Thirty two subsets of the global model were constructed to represent combinations of factors believed to influence brood habitat use (Table 1). Therefore, 32 models were assessed on how well they fit the data using Akaike's Information Criterion (AIC; Burnham and Anderson 2002). Akaike's Information Criterion (AIC) is an extension of likelihood theory, and AIC is a relative distance between model pairs (Burnham and Anderson 2002). The distance is related to the Kullback-Leibler distance of information theory (Burnham and Anderson 2002). This distance is a measure of entropy (i.e. information lost) for models used to approximate truth (Burnham and Anderson 2002). We modified the AIC values because of low sample sizes to AIC_C (Burnham and Anderson 2002). The relative fit of each candidate model was assessed by calculating Akaike weights (Burnham and Anderson 2002), weighting of models with a value from 0 to 1, with the best-fitting candidate model having the greatest Akaike weight. All models were assessed for goodness of fit using Hosmer–Lemeshow tests (models with p-values $\geq$0.05 did not fit the data) using the lack-fit details option in SAS. We analyzed all data using SAS (SAS Institute, Inc. 2003).

Results

During 2005, we obtained 58 brood locations for 5 unique broods. These were the only broods that had chicks remaining after 14 days. The average MCP home range size of broods was 5.53±2.43 ha. We evaluated a total of 51 locations for both random-within and random-outside points for a total of 102 random locations. The most common vegetation type found among individual habitat parameters was sod-forming grasses in both brood and random-within locations (Table 3). Bunchgrasses and forbs also were major components at all 3 location types. We observed small amounts of blackberry during sampling; however, it was frequently observed at brood locations when available. Broad-leaf woody vegetation and legumes also were found more frequently at brood locations than random-within and random-out locations. More grass-litter was detected at random-within locations than brood and random-out locations. Bare ground, litter, or broad-leaf woody vegetation were not significantly different among the 3 location types (Table 3). Stepwise logistic regression retained all of the habitat parameters at both levels of analysis except blackberry and litter (Tables 4, 5).

We examined 16 hypothesized models using 58

Table 4: Significant predictors of probability of northern bobwhite brood use in Florida, USA 2005, based on a stepwise logistic regression model contrasting habitat measured at used locations and random locations within MCP Home-range. See Table 2 for parameter descriptions.

Parameter	DF	Estimate	Standard Error	Wald Chi-Square	$Pr > \chi^2$
Intercept	1	-17.0223	3.6527	21.7177	<0.0001
sofg	1	8.7960	2.1073	17.4231	<0.0001
forb	1	9.3797	2.3936	15.3564	<0.0001
bugr	1	6.9192	1.6978	16.6085	<0.0001
blwd	1	6.3663	1.8276	12.1335	0.0005
legm	1	9.1853	2.9079	9.9777	0.0016
bare	1	9.0291	2.5216	12.8214	0.0003

brood locations and 51 random locations within the MCP home range. The best approximating model (w_1=0.94) for predicting brood versus random-within locations included all significant habitat variables: bird identification (a blocking variable), sod-forming grasses, forbs, bunchgrasses, broadleaf-woody legumes, and bare ground (Table 6). All other models were poor at predicting brood use; no competing models were within 2 AIC_C of the best model.

The best approximating model (w_1=0.94) for predicting brood versus random-outside locations included all significant variables except bird identification (Table 7). All other models were poor at predicting brood use; no competing models were within 2 AIC_C of the best model.

Discussion

These results should be interpreted with caution because our study suffered from a small sample size, limiting our ability to draw upon conclusions from our results with high statistical confidence. We believe, however, that the data and results presented elucidate, or minimally highlight, some interesting occurrences regarding brood habitat use in pastureland.

Interpretation of our data suggests, while based on small sample size, that at the microhabitat scale no single vegetation type can be used to predict use of habitat by bobwhite broods. This may indicate that areas with a variety of microhabitat (i.e. habitat diversity is high) characteristics favor brood use. This observation is consistent with Burger et al. (1993); they found that optimal brood-rearing habitat should contain high plant species richness favoring forbs. Yates et al. (1995) also found mosaic-type land cover beneficial to broods.

Sod-forming grass was a major component of habitat at brood locations and is an anomaly in terms of bobwhite ecology. Dense vegetation has been found to impede chick mobility (DeVos and Mueller 1993), as well as act as a fatal heat trap (Burkhart 2004). We believe a couple mechanisms potentially caused this result: (1) the ranch is dominated by Bahia grass pastures comprised mainly of sod-forming grasses making it so available and virtually unavoidable by brooding bobwhites; and (2) because of the low mobility of broods, the small patches of other types of vegetation available are not generally accessible by broods. A reduction in cattle grazing on the study area in July 2005, as a result of ownership change, may have resulted in higher percentages of sod-forming grasses at bird locations than other years.

We also found broad-leaved woody vegetation, forbs and legumes to be more prevalent at brood locations than at random-within and random-out loca-

Table 5: Significant predictors of probability of northern bobwhite brood use in Florida, USA, 2005, based on a stepwise logistic regression model contrasting habitat measured at used locations and random locations outside MCP home range. See Table 2 for parameter descriptions.

Parameter	DF	Estimate	Standard Error	Wald Chi-Square	$Pr > \chi^2$
Intercept	1	-20.4338	4.5853	19.8588	<0.0001
sofg	1	14.1201	3.3034	18.2706	<0.0001
forb	1	7.0954	2.126	11.1384	0.0008
bugr	1	11.2451	2.9462	14.5685	0.0001
blwd	1	7.5642	2.5648	8.6982	0.0032
legm	1	13.5051	5.1516	6.8725	0.0088
bare	1	11.1869	3.2356	11.9537	0.0005

tions. Jackson et al. (1987) found that brood-rearing habitat was most dependent on invertebrate abundance. During the first 2-3 weeks post-hatching, bobwhite chicks consume >80% invertebrates that provide essential nutrients for growth and survival (Handley 1931, Nestler 1940). Broad-leaved woody vegetation, forbs, and legumes provide good habitat for insects, and thus an abundant food supply for bobwhite chicks. The structure of broad-leaved woody vegetation also provides dense cover from rain and avian predation, further increasing the probability of brood survival. Broods also selected habitats with a higher woody component for roosting and possible escape cover from predators (Johnson and Guthery 1988).

Bunchgrass was also found more frequently at brood locations than at random-within locations. Bunchgrasses provide excellent nesting habitat but can impede brood mobility when stands are too dense. This further exemplifies the impact of spatial scale on broods-hens often choose to nest in bunchgrass because of its benefits regardless of the effect it can have on broods. Post-hatching, brood mobility is limited and may be further impeded by bunchgrass if a hen chooses to nest in it. The diversity of plants within bunchgrass patches creates a more suitable environment than a solid bunchgrass stand.

We believe the key component is spatial scale of landscape compared to mobility of broods. Bobwhites select habitats at many spatial scales (James Martin, unpublished data). Throughout their range they prefer early seral stages of habitat, and within those habitats bobwhites prefer a of diversity microhabitats. However, bobwhites have relatively poor dispersal and mobility-limiting their ability to occupy more suitable sites when large distances from their hatch site. Cook (2004) found that one-fourth to one-third of bobwhites in southern Georgia dispersed up to nearly 2,200 m prior to the breeding season. The remaining bobwhites retained a home range in the same area as their brood home range. More broadly, two-thirds to three-quarters of bobwhites remain in habitat that is spatially close to or the same as the habitat they inhabited as chicks. Consequently, they are confined to that habitat into which they hatch. Comparing brood locations to random-within locations therefore reveals much about the preferred vegetation for brood habitat use at the fourth-order scale (Johnson 1980). However, these data reveal little to how bobwhites are affected at larger spatial and temporal scales.

The combination of variables (i.e. diverse habitat) included in the best approximating model for predicting brood versus random-within locations favor brood use. Our data suggest that a diversity of vegetative type among canopy coverage is crit-

Table 6: Logistic Regression Models predicting brood locations (n = 58) versus random within MCP home range (n = 51) using ground cover data collect in Florida, USA, 2005. See Table 2 for variable descriptions.

		Hosmer-Lemeshow					
Model	DF	χ^2	P-Value	K	AIC_c	Δ	w_i
intercept + birdid + sofg + forb + bugr + blwd + legm + bare	8	6.06	0.64	8	87.33	0.00	0.95
intercept + sofg + forb + bugr + blwd + legm + bare	8	14.19	0.08	7	93.05	5.72	0.05
intercept + birdid + sofg + forb + bugr + legm + bare	8	6.75	0.56	7	102.64	15.32	0
intercept + birdid + sofg + forb + bugr	8	4.88	0.77	5	107.81	20.48	0
intercept + sofg + bugr + blwd + legm + bare	8	8.82	0.36	6	115.20	27.87	0
intercept + sofg + bugr + blwd + bare	8	7.13	0.52	5	119.42	32.09	0
intercept + birdid + sofg + bugr	8	4.88	0.77	4	121.89	34.56	0
intercept + sofg + bugr	8	8.19	0.42	3	127.45	40.12	0
intercept + birdid + legm	6	12.04	0.06	3	144.69	57.36	0
intercept + bugr	6	13.53	0.04	2	148.88	61.55	0
intercept + bugr + blwd	7	16.31	0.02	3	149.10	61.77	0
intercept + birdid + sofg	8	9.23	0.32	3	149.91	62.59	0
intercept + blwd	4	1.97	0.74	2	150.59	63.27	0
intercept + birdid + bugr + blwd	8	5.45	0.60	4	152.86	65.54	0
intercept + birdid + bugr	8	4.41	0.73	3	154.15	66.82	0
intercept + birdid + blwd	8	4.45	0.73	3	157.61	70.28	0

Table 7: Logistic Regression Models predicting brood locations (n = 58) versus random outside MCP homerange (n = 51) using ground cover data collected in Florida, USA, 2005. Model, df, Hosmer-Lemeshow goodness-of-fit statics, number of parameters (K), AIC_c, Δ, and w_i values are presented.

		Hosmer-Lemeshow					
Model	DF	χ^2	P-Value	K	AIC_c	Δ	w_i
intercept + sofg + forb + bugr + blwd + legm + bare	8	3.79	0.80	7	59.46	0.00	0.95
intercept + birdid + sofg + forb + bugr + blwd + legm + bare	8	4.40	0.82	8	65.30	5.83	0.05
intercept + birdid + sofg + forb + bugr + legm + bare	8	12.94	0.11	7	72.22	12.75	0
intercept + sofg + bugr + blwd + legm + bare	8	4.60	0.80	6	73.15	13.68	0
intercept + sofg + bugr + blwd + bare	8	4.18	0.84	5	80.04	20.58	0
intercept + sofg + bugr	7	5.86	0.56	3	107.92	48.46	0
intercept + birdid + sofg + forb + bugr	8	4.79	0.78	5	111.60	52.13	0
intercept + birdid + sofg + bugr	8	7.24	0.51	4	111.68	52.22	0
intercept + birdid + sofg	8	5.06	0.75	3	125.02	65.56	0
intercept + birdid + legm	6	5.87	0.44	3	128.03	68.57	0
intercept + blwd	5	10.39	0.06	2	148.75	89.28	0
intercept + bugr + blwd	8	25.38	0.00	3	150.54	91.08	0
intercept + bugr	6	25.58	0.00	2	152.58	93.11	0
intercept + birdid + blwd	7	6.81	0.45	3	155.46	96.00	0
intercept + birdid + bugr + blwd	8	15.43	0.05	4	156.84	97.38	0
intercept + birdid + bugr	7	14.15	0.05	3	159.77	100.31	0

ical to brood use. Past bobwhite research agrees with this conjecture because bobwhites favor early-successional habitat which is diverse in terms of both canopy structure and plant community.

Management Implications

The management implications of this study are limited because of our lack of sample size and the inability to draw conclusions from the data with high statistical precision. However, our results suggest that managing habitat to create a diverse plant community will increase the probability of use by bobwhite broods. Furthermore, usage of these habitats by broods may increase chances for survival; future studies incorporating brood use into survival estimation models is needed to examine how habitats used affect chick survival. Managing habitat across large scales (>1000 ha) will improve accessibility to favorable habitats for entire populations of bobwhites, but managers should not overlook fine-scale habitat management, particularly regarding that of brood habitats, to improve conditions for bobwhite chicks.

Future research with larger sample sizes is warranted to substantiate our results. Also, it would be helpful for researchers to examine how habitat(s) used may affect the success of the brood itself (i.e. chick survival).

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Conservation

Managing Black-throated Bobwhite for Sustainability in Belize: Preliminary Results of a Population Study

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The Black-throated Bobwhite (*Colinus nigrogularis*), a close relative of the Northern Bobwhite (*C. virginianus*), has a distinctive black throat and eye-stripes with both bounded by white. Black-throated Bobwhites occur in three distinct geographically isolated populations. Currently no limits or seasons are in place and a hunting license is the only requirement for harvesting this species in Belize. Little is known about Black-throated Bobwhite populations in Belize and data on the impact of hunting on this species is lacking. Because of its restricted distribution, it was recommended that Black-throated Bobwhites should be removed from the list of legally hunted species pending a better understanding of its population dynamics. In 2006 we initiated a long-term study of Black-throated Bobwhite biology at the 469 km^2 Manatee Forest Reserve (henceforth MFR). Data collected at the end of the wet season indicated a population of 0.072-0.144 quail per hectare, which was appreciably lower than the density reported in the heneguen growing region of Yucatan, Mexico. Such densities predictably decreased throughout the dry season (breeding season).

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Key words: black-throated bobwhite, Belize, *Colinus nigrogularis*, population estimate

Introduction

The Black-throated Bobwhite (*Colinus nigrogularis*) is a close relative of the Northern Bobwhite (*C. virginianus*) but is easily distinguished from the Northern Bobwhite by a distinctive black throat and eye-stripes, both bounded by white (Johnsgard 1988). Black-throated Bobwhites occur in three distinct geographically isolated populations (Figure 1; Johnsgard 1988). Both *C. n. caboti* and *C. n. persiccus*, inhabit grasslands within the states of Campeche, Quintana Roo, and Yucatan, Mexico (Ornat et al. 1989). *C. n. nigrogularis* inhabits pine coastal savannas of central Belize and eastern Guatemala (Jones and Vallely 2001) and *C. n. segoviensis* occurs in pine savannas of the Mosquitia on the Honduran/Nicaraguan border (Howell 1971, Roberto Gallardo personal communication).

Under the Wildlife Protection Act of 1981, the Black-throated Bobwhite is one of six bird species legally hunted in Belize. Currently no limits or seasons are in place and a hunting license is the only requirement for harvesting this species. Little is known about Black-throated Bobwhite populations in Belize and data on the impact of hunting on this species are lacking. Because of its restricted distribution, Miller and Miller (1997) recommended that Black-throated Bobwhites should be removed from the list of legally hunted species pending a better understanding of its population dynamics.

Study Area

In 2006 we initiated a long-term study of Black-throated Bobwhite population biology at the 469 km^2 Manatee Forest Reserve (henceforth MFR). The MFR (N 16° 40.8′ W 0880 25° 34.8′) is in the Tropical Moist Forest Life Zone (Holdridge 1967, Figure 1, Figure 2) of Belize. Forests on this coastal

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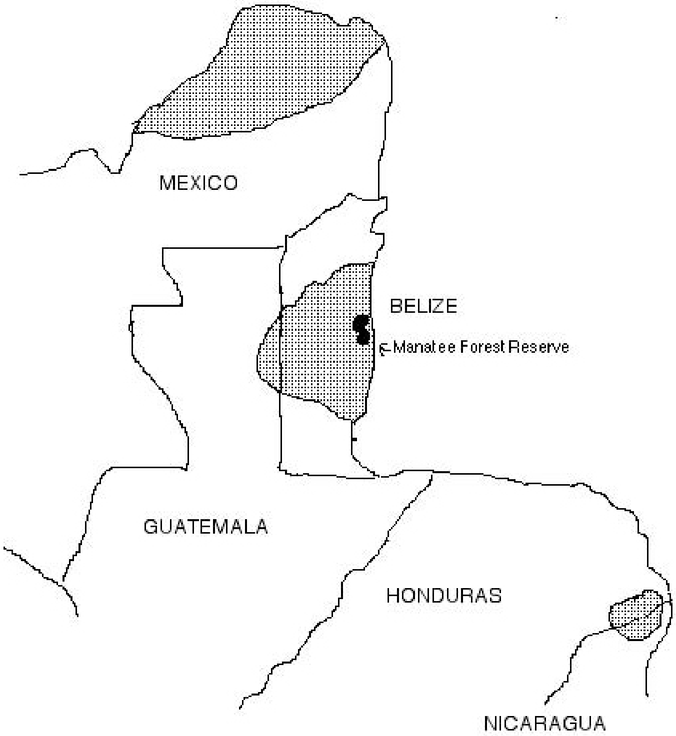

Figure 1: Distribution of Black-throated Bobwhite and location of Manatee Forest Reserve.

plain consist of, a mosaic of species predominantly determined by soil type, drainage and fire. While broadleaf forests occur near rivers, nearby soil fertility and poor drainage result in a range of forest types from mixed pine-broadleaf forest to sparse pine-palm, and grassland savanna (Johnson and Chaffey 1974). Rainfall in the central region of Belize is seasonal, generally receiving 2000-2700 mm annually, with less than 100 mm/month during the dry season, January through May (Walker 1973).

Methods

Field Procedures

For our preliminary sampling, we randomly established four stratified sampling grids in the MFR on the basis of accessibility. Grids consisted of five 500-m lines spaced 300 m apart. Numbered stakes and GPS readings delineated beginning, middle and end points of each line. Each quadrant is described by vegetative composition (Table 1). We estimated density by taking the largest number of quail observed during any single survey and divided by the total area of each quadrant. We maintained a 150-m observational buffer around each quadrant. This protocol allowed us to estimate relative population size and detect seasonal changes in the Black-throated Bobwhite quail population density.

Once during the end of the wet season (January) and twice monthly during the dry season an observer walked each transect, within each quadrant, and recorded all quail visually observed. Because the grasses are of the "bunch grass" variety, with open spaces between bunches, we feel confident the majority of quail within the buffer area were noted. Auditory counts were not included (unless the bird was visually located) because of high observer variability and subsequent bias. For each quail observed, the location, number of quail

Figure 2: Black-throated Bowhites walking along dirt road in Manatee Forest Reserve, Belize during 2006.

observed, and the distance and angle to the point where a quail was initially sighted were recorded. We used a laser rangefinder to determine distances to observed quail and a compass to determine the angle from the transect. Start times and weather conditions were also recorded. To avoid temporal bias, each survey was conducted in a reverse order from the previous survey. For example, quadrants 1, 2, 3, 4, were sampled and then the sequence was reversed (4, 3, 2, 1) within the quadrant. The observer also reversed the direction walked (i.e., traveling line 1, 2, 3, 4 then reversing the order at the next survey). We also used the amount of time (minutes) required to record the total number of observations (total amount of time spent/number of quail observed) averaged from March-June as a measure of unit effort.

Table 1: Black-throated Bobwhite observations by month during January-June 2006 in Manatee Forest Reserve, Belize.

Month	Total Observations	Total # Quail	Average quail/observation
January	4	36	9
February	0	0	0
March	5	6	1.2
April	7	15	2.1
May	16	24	1.5
June	5	10	2
Average	7.4	13.75	1.7

Table 2: Four quadrants of the Manatee Forest Reserve, Belize with habitat descriptions during 2006 surveys.

Quadrant	Quail/Ha.	Quadrant Description
Quadrant 1 UTM 1911667	0.144	Northern portion Caribbean pine, pricky plants and shrubs. Central area savanna. South end swampy with tall grass during dry season. Area burned in 2004
Quadrant 2 UTM 1910141	0.112	Western area open savanna with patches of Palmetto palm. Gallery forest runs lengthwise in the central area. Eastern section is open savanna with scattered Caribbean pine. Eastern edge dense shrub.
Quadrant 3 UTM 1910836	0.08	Western edge Caribbean pine with shrubby undergrowth, patches of open grass. Eastern section fewer pines with more shrub and grasses. Selectively logged in 2004 and burned in 2005.
Quadrant 4 UTM 1911605	0.072	Western area Caribbean pine with shrubby, prickly undergrowth. From central area to eastern area pine is mixed with oak. More dense undergrowth than other quadrants. Selectively logged in 2004 and burned in 2005

Results

Preliminary density estimates of Black-throated Bobwhite, quadrant location, and vegetative components were determined for the first season of monitoring (Table 2). While the number of observations (4.0 versus average 7.4) in January (end of wet season/beginning of the dry season) was low the total number of quail was greatest (36.0 versus average 13.75) during this period (compared to the remainder of the dry season). While the total number of observations of quail showed an increasing linear trend, throughout the dry season, the number of quail, per observation, varied across all periods.

Discussion

Little has been published on the population status of this species (Table 3). Klass (1968) reported that during June a henequen (*Agave fourcroydes*) field near Merida, Yucatan had a quail density between 2-4 birds per hectare. Tramer (1974) worked in an area,

Table 3: Overview of Black-throated Bobwhite density estimates from Mexico and Central America.

Location	**Density (Quail/ha)**	**Source**
Yucatan, Mexico	**4.11/ha**	**Tramer 1974**
Yucatan, Mexico	**2-4/ha**	**Klass 1968**
Manatee FR, Belize	**0.1/ha**	**Eitniear et al. This paper**
Waspam, Nicaragua	**0.02/ha (0.01 territories/ha)**	**Howell 1971**

Figure 3: Habitat conditions within Manatee Forest Reserve, Belize after 13 April 2006 burn.

Figure 4: Flooding in Manatee Forest Reserve, Belize just prior to June 2006 scheduled survey forcing it to be cancelled.

during the dry season, near henequen fields and calculated a density of 2.43 quail per hectare. However, monoculture agricultural fields like henequen, while proving abundant food resources and protection from predators (the plant's leaves are tipped with spines), may not be representative of habitat throughout the remainder of the species range. In Nicaragua, Howell (1971) visited areas suitable for Black-throated Bobwhite throughout the year. While avoiding areas recently burned he determined the number of quail territories to be 0.012 per hectare (0.02 quail per hectare). Our preliminary estimate of 0.1 quail per hectare represents what we believe is more typical of densities found in suitable habitat. The greater quail density reported by Howell (1971) may be the result of a more robust sample size.

Several factors may have both immediate and long-term effects on quail populations at our sites. While the coastal plain has been shaped by fire, the immediate impact on quail numbers due to fire is unknown. Undoubtedly the long-term impact on local quail populations will be the result of how fire shapes vegetative communities. Such changes will likely depend on the diversity of plants at the site prior to the fire and the frequency of fires. Anecdotal evidence suggests that areas adjacent to the reserve are burned frequently (every 1-2 years) to promote new growth of grasses that attract deer, which are then hunted. Areas that sustain annual or biannual fires have evolved into grassland with few trees and a sparse shrub layer. Within the MFR all quadrants have been documented to have burned, at least partially, in the past three years. Two events during data collection in 2006 should be considered when interpreting our results. On 24 April a fire burned about 50% of quadrant 1 and on 13 April a fire burned a strip of vegetation from the northwest to southeastern corner of quadrant 2. Given that data were collected on these quadrants on 13 and 27 April the possible impact of a recent burn on local quail populations should be considered when interpreting survey data (Figure 3). Finally, a second survey in June was cancelled because of the onset of the rainy season (Figure 4). These various factors make maintaining surveying a challenge.

Conclusions

Preliminary results support statements by Leopold (1972) that Black-throated Bobwhite quail exist in coveys during the wet season breaking up into smaller groups for breeding in the dry season. Data collected at the end of the wet season (January) indicated a population of 0.072-0.144 quail per hectare which was appreciably smaller than the density reported in the henequen growing region of Yucatan, Mexico. Such densities predictably decreased throughout the dry season (breeding period). Flooding, and to a lesser extent fires, are a seasonal occurrence. Their immediate impact on quail numbers needs to be explored.

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Delineation of Gambel's Quail Habitat in the Trans-Pecos, Texas

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Gambel's quail (*Callipepla gambelii*) are a common quail species in southwestern states of the United States and the northwestern states of Mexico. In Texas this species occurs in the Trans-Pecos region, are underutilized, and could become an important game bird and source of income for ranchers in the Chihuahuan Desert region of Texas. Salt cedar (*Tamarisk* spp.), introduced from Asia for ornamental and erosion purposes, is invasive in the western part of the Rio Grande corridor that generally creates monocultures that choke out the native vegetation of the region. Knowing this, the objectives of this study were to: (1) delineate salt cedar and native riparian habitats along the Rio Grande corridor in the Trans-Pecos; (2) evaluate those habitats based on the known distribution of Gambel's quail in the Trans-Pecos; and (3) estimate the amount of suitable habitat for Gambel's quail in Trans-Pecos, Texas. Although dominant along the Rio Grande, native riparian vegetation was more prevalent than salt cedar communities when combining primary creeks in all counties. Brewster County was the area with a higher percentage of salt cedar occurrence (21.2%) vs. native riparian vegetation (78.8%). The largest extension of salt cedar occurred in Presidio County with an extension of 6,656.3 ha but this only represented 12.7% of our analyzed area. Hudspeth County had an occurrence of salt cedar of 2,905.2 ha representing 6.8% of the estimated riparian area of the Rio Grande corridor in this county. El Paso County's total urban area-agricultural fields area is a total of 90,682.9184 ha.

Citation: Ortega-Sanchez A, Harveson LA, Lopez RR, Sullins MR. 2009. Delineation of Gambel's quail habitat in the Trans-Pecos, Texas. Pages 106 - 110 in Cederbaum SB, Faircloth BC, Terhune TM, Thompson JJ, Carroll JP, eds. Gamebird 2006: Quail VI and Perdix XII. 31 May - 4 June 2006. Warnell School of Forestry and Natural Resources, Athens, GA, USA.

Key words: *Callipepla gambelii*, Gambel's quail, GIS, habitat, riparian vegetation, salt cedar, Texas, Trans-Pecos

Introduction

Gambel's quail (*Callipepla gambelii*) are a common quail species in the southwestern United States and northwestern states of Mexico which in Texas, occurs exclusively in the Trans-Pecos region (Oberholser 1974, Brown et al. 1998). In Texas Gambel's quail are a game bird and due to populations declines of bobwhite quail (*Colinus virginianus*) (Brennan 1991, 2002, Peterson et al. 2002) could potentially increase in importance as a game bird and serve as an additional source of income for ranchers in the Chihuahuan Desert of Texas.

In the Trans-Pecos region Gambel's quail show preference for riparian vegetation (Gray 2005). Salt cedar (*Tamarisk* spp.), a species introduced from Asia as an ornamental plant and for erosion control, has become a dominant vegetative component of riparian systems along the western part of the Rio Grande corridor (Everitt et al. 1996) and now occupies approximately 460 km of the river corridor (Everitt et al. 2006). Several studies have reported that Gambel's quail show a preference for native riparian vegetation over invasive salt cedar thickets (Engel-Wilson and Ohmart 1978, Gray 2005) and subsequently the objectives of this study were to: (1) delineate salt cedar and native riparian habitats along the Rio Grande corridor in the Trans-Pecos; (2) evaluate those habitats based on the known distribution of Gambel's quail in the Trans-Pecos; and (3) estimate the amount of suitable riparian habitat for Gambel's quail in Trans-Pecos, Texas. This information could aid resource managers in the Trans-Pecos in managing habitats for Gambel's quail in Texas.

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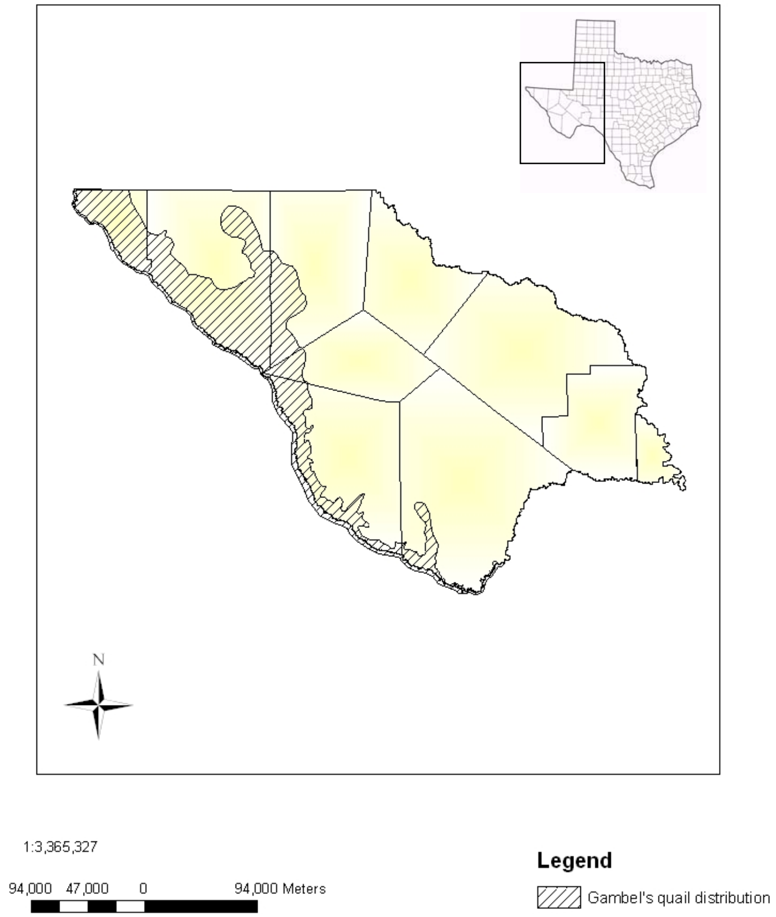

Figure 1: Gambels quail distribution consists of the Rio Grande corridor and adjacent draws and arroyos in Trans-Pecos, Texas.

Study Area

Our study area encompasses the Trans-Pecos region of Texas where we restricted our analysis to the most recent distribution area for Gambel's quail according to Sullins (2006) and the Texas Parks and Wildlife Department (Figure 1). Powell (1998) described the native vegetation in this region as Chihuahuan Desert Scrub, which is present at lower elevations (1000-1150)with precipitation ranging from 18 to 31 cm/year and dominated by shrub species such as creosote (*Larrea tridentata*) and semi-succulents such as lechuguilla (*Agave lechuguilla*), sotol (*Dasylirion* spp.), and yucca (*Yucca* spp.).

Methods

To delineate native and exotic riparian habitats for our study site, we used Digital Ortho Quarter Quadrangles (DOQQs [1-m resolution]) obtained from the Texas Natural Resource Information System (TNRIS [http://www.tnris.state.tx.us]). When delineating salt cedar using remote sensing techniques, Everitt et al. (1996) recommended using imagery obtained in fall-winter. During this time, salt cedar provides a unique signature because of the coloration of its foliage being different from that of native vegetation. based on this we used from the fall of 1995 corresponding to an area within 1-km from the Rio Grande. In the event of a missing DOQQ

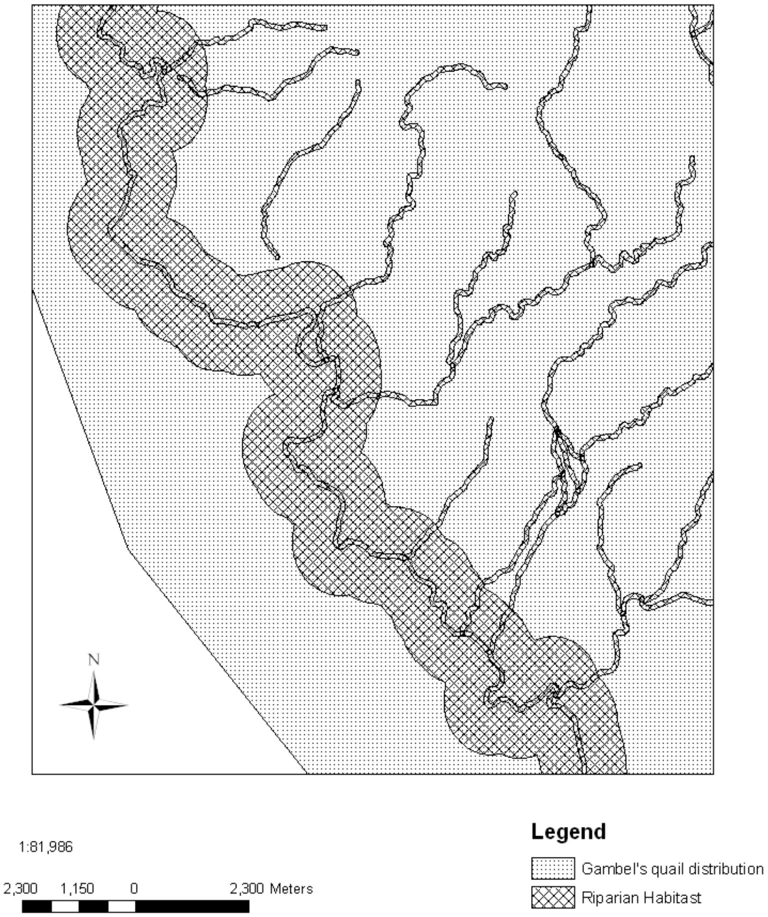

Figure 2: The Rio Grande corridor was represented by a 1,000-m buffer and adjacent draws and arroyos were represented by a 50-m buffer.

from 1995 we substituted 2004 images for the analysis.

Using the Mosaic Tool from the ERDAS Imagine 9.0© software (Leica Geosystems, GIS & Mapping, LLC) we mosaiced the DOQQs for the counties of Brewster, Presidio, El Paso, and Hudspeth (Jeff Davis and Culberson Counties were included with Hudspeth County because of the small amount of DOQQs required for their area). The 4 resultant mosaics were classified with unsupervised classification under the classifier function in ERDAS into 3 classes; native riparian, salt cedar, and scrubland. Scrubland was described as any area within Gambel's quail distribution that did no fall into the riparian habitat buffer zones. We obtained medium resolution hydrography files from the National Hydrography Dataset from U. S. Geography (http://nhdgeo.usgs.gov/). As recommended by Gray (2005) we buffered all flow-line shapefiles within the Gambel's quail distribution map using a 50-m buffer in ArcGIS 9.0 ©(ESRI, Inc., Redland, CA). Similarly, the Rio Grande corridor was assigned a 1,000-m buffer (Figure 2) which encompassed most riparian habitats (Gray 2005).

The classified mosaics were then clipped to the river buffer. Zonal statistics function from Spatial Analyst (ESRI, Inc., Redland, CA) package was run for the classification clipped to the river buffer to de-

Table 1: Total area and occurrence (%) of habitats delineated from an unsupervised classification of DOQQs imagery within Gambels quail distribution in the Trans-Pecos, Texas, 2005.

	Habitat Class								
	Scrubland		Native riparian		Salt cedar		Urban/Ag.		Total
County	ha	% area	ha	% area	ha	% area	ha	% area	ha
Brewster County	66,503	75.9	16,623	18.9	4,490	5.2			87,615
Presidio County	140,770	73.1	45,516	24.1	6,656	2.6			192,942
Hudspeth County*	162,321	80.2	39,811	19.4	2,905	1.4			205,037
El Paso County	70,733	42.7					90,683	57.3	165,416

*Jeff Davis and Culberson Counties included

termine area. Although most Gambel's quail habitat in the Trans-Pecos has not been affected by urban sprawl, much habitat has been lost in El Paso County (Harveson 2007), subsequently for El Paso County, we additionally delineated cover classes to include agriculture fields and urbanized areas. However, because of the complexity and similarity in color, shadow, and texture of these 2 habitats they were combined to form 1 cover class (urban-agriculture).

Results

Within the distribution of Gambel's quail in Texas, native riparian habitat was predominant in riparian corridors compared to salt cedar within all counties (Table 1). Brewster County contained the highest percentage of salt cedar (21.2%), followed by etc, (Table 1). The contiguous area of salt cedar occurred in Presidio County with 6,656.3 ha and Hudspeth County 2,905.2 ha representing 12.7% and 6.8% of the estimated riparian area of the Rio Grande corridor. El Paso County's urban-agriculture area totaled 90,682.9 ha, comprising 57.3% of the potential Gambel's quail habitat in the county. Brewster County presents the highest salt cedar:native riparian habitat 1:3.5 ratio. The river corridor that borders Brewster County is relatively smaller compared to what borders Presidio and Hudspeth Counties.

Discussion

Engel-Wilson and Ohmart (1978) and Gray (2005) reported that Gambel's quail prefer native riparian habitat; this could be an influencing factor for Gambel's quail not to expand their distribution in this county even in good years as were 2004 and 2005. The extended drought that the Trans-Pecos has had for the past 10 years (Simpson 2005) could have affected the dispersion of salt cedar and Gambel's quail for this county. The salt cedar habitat in both Presidio and Hudspeth Counties was less than native riparian habitat. But the areas that these 2 counties represent are larger than the area in Brewster County which is similar to the findings of Everitt et al. (1996). Because of El Paso County's complex urbanization format it is very difficult to distinguish classes between urban areas and agriculture from aerial photography. From observation of the DOQQs and personal communication with people familiar to the area, the pattern that these areas have is plantings of different crops with occasional salt cedar wind barriers. In our study we decided to report the entire urbanized area along with the agricultural fields that and the total area of the county. We decided to report this information because the distribution map we used showed that the entire county formed part of Gambel's quail distribution.

Management Implications

Further studies need to be conducted on the impacts of salt cedar on Gambel's quail. Additionally, the distribution of Gambel's quail has declined in Brewster County, where salt cedar has increased. Salt cedar encroachment along with other factors could be affecting the existing range of Gambel's quail. Salt cedar should be controlled with brush management practices in order to increase suitable habitat for Gambel's quail. Further research should be conducted on alternatives to separate salt cedar from agricultural fields and urban areas to better refine our delineation of cover classes in El Paso County. Further, a better understanding of the impact urbanization and fragmentation has on quail populations is needed for the Rio Grande corridor. Although Gambel's quail are known to be present in these areas, there is a lack of information on the behavior and movements of Gambel's quail in these habitat types. Knowing the areas with higher probability of Gambel's quail occurrence will aid ranchers and natural resource managers to direct their management practices towards these target areas. Considering salt cedar is a possible threat to suitable quail habitat, control practices we believe should be applied for its' control.

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Habitat Use and Survival of the Spotted Tinamou (*Nothura maculosa*) in Agroecosystems in the Province of Buenos Aires, Argentina

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Changes in the composition and configuration of agricultural landscapes stemming from grassland conversion and agricultural intensification have contributed to the global declines of many grassland and shrubland birds. In both North America and Europe there exists a large body of research on the effects of agriculture on populations of terrestrial gamebirds. However, little research exists for these species in Argentina or Latin America in general. In Argentina the most important gamebird species is the spotted tinamou (*Nothura maculosa*). This species has become increasingly scarce in a significant portion of its range, possibly due to agricultural intensification over the last 15 years. Using radio telemetry, we examined habitat use, movements, and survival of spotted tinamous in 2 landscapes in the province of Buenos Aires, Argentina; one dominated by annual row crops and the other used for annual crops and grazing. During winter, individuals used in order of preference: fallow fields and areas with short herbaceous vegetation, followed by wetlands. Areas in winter wheat and field edges were used least in relation to their availabilty. Although birds generally maintained small home ranges, in some cases changes in cattle density and the structure of row crops caused birds to move considerable distances. Survival mid-winter to early spring was more than double in the mixed landscape ($\hat{s}$ = 0.73, SE = 0.19) compared with the landscape dedicated to row crops ($\hat{s}$ = 0.33, SE = 0.19). Considering how research in other parts of the world has demonstrated the effects of agricultural intensification on terrestrial gamebirds, these results are not unexpected and suggest a precarious future for the conservation of grassland and agroecosystem species in Argentina in light of present agricultural trends.

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Key words: agricultural intensification, conservation, *Nothura maculosa*, spotted tinamou, survival

Introduction

Globally, populations of grassland and shrubland birds have been declining due to habitat conversion and agricultural intensification (Askins 2000, Goriup 1988, Murphy 2003, Pain and Pienkowski 1997, Vickery and Herkert 1995). In agroecosystems of austral South America habitat loss and the intensification in management have been extensive and rapid, particularly in the pampas of Argentina starting in the early 1990's, typified by the increased use of external inputs, increased yields, and a shift towards agricultural production for export markets (Ferreyra 2001, Ghersa et al. 2002, Hall et al. 2001, Solbrig and Vera 2001, Viglizzo et al. 2001).

The spotted tinamou (*Nothura maculosa*) is a common bird of grasslands and agroecosystems in eastern austral South America, one of the most important terrestrial gamebirds in the region, and formerly common in agricultural systems (Bucher and Nores 1988, Bump and Bump 1969, Cabot 1992, Davies 2002, Menegheti 1985). In recent years, within the pampas of Argentina, the spotted tinamou has become increasingly conspicuous by their absence apparently stemming from the expansion and intensification of grazing and row crop practices.

All tinamous are relatively poorly studied; however, in austral South American grasslands the tina-

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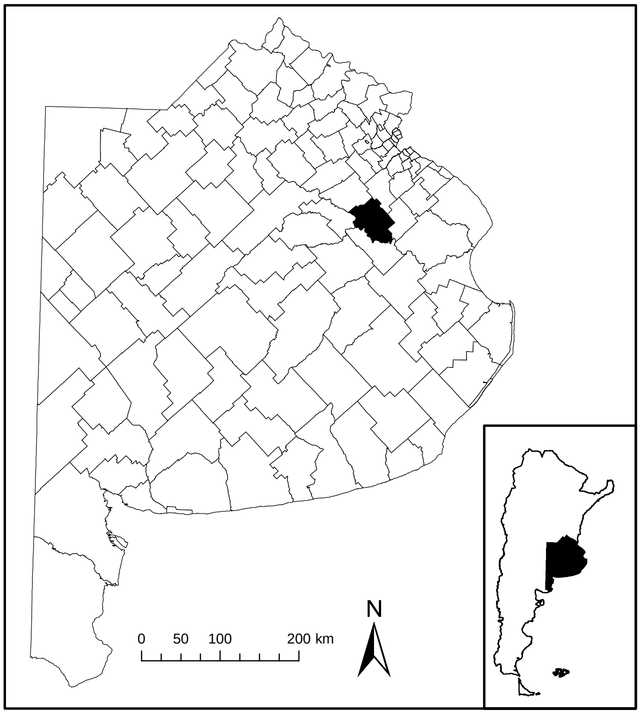

Figure 1: Map showing the location of the district of San Miguel del Monte in the Province of Buenos Aires, Argentina

mous replace the Galliformes and are their ecological equivalent, which allows for inferences to be drawn among the Galliformes and the Tinamiformes in regard to tinamou ecology (Thompson 2004). We used radio telemetry to investigate our theory of ecological equivalence. Based upon the observed effects of agricultural intensification on Galliformes and existing knowledge of the spotted tinamou that within pampean ecosystems we predicted that survival of spotted tinamous would be negatively correlated, and home range size positively correlated, with increasing land use intensity while habitat selection would favor areas most similar to natural grassland in vegetative structure (Bump and Bump 1969, Thompson 2004).

Study Area

Our study sites were located in the district of San Miguel del Monte in the province of Buenos Aires, Argentina (Figure 1). San Miguel del Monte is located in the flooding pampa, a regional subdivision of the ~760,000 km^2 Río de la Plata grassland system that covers northeastern Argentina, Uruguay and southeastern Brazil (Soriano et al. 1991). Traditionally the flooding pampa has been used principally for extensive livestock production (Hall et al. 1988), however, since the early 1990's row crop agriculture has become an increasingly important land use.

We selected two study sites; one dedicated to row crops and the other used for used for a mix of row crops and grazing. The row crop site was 160 hectares, of which 85% was used for soybean, corn, and winter wheat production, and the remaining area comprised of wetlands or field borders. The site with mixed row crop and grazing uses was 230 hectares, 50% of its area used for soybean, corn, and winter wheat production, and the remainder, including wetlands, used for cattle grazing.

Methods

During July 2003, we fitted 4 birds with pendant-style transmitters (6.0 g, 2.2-2.3% of body mass) equipped with an activity switch (Holohil Systems Ltd., Ontario, Canada) at the row crop dominated site and 14 birds in June 2004 at the mixed use site. In 2004, no birds were radio-tagged at the row crop site because none were detected over a 2 month search in the autumn of that year. All birds were captured at night using spotlights and hand nets. Due to uncertainties in sexing birds related to age (Bump and Bump 1969), sexual differences were not included in the study. In both years birds were located 3 times per week from the date of capture until October 23 (mid-winter to early spring) dependent upon accessibility to the sites.

Due to mortality, insufficient number of radio locations, radio failure, or radio loss we used 3 birds from the row crop site and 8 from the mixed use site in our analysis. Locations were entered into a geographic information system (GIS) for each site using ArcGIS software (Environmental Systems Research Institute, Inc.). Minimum convex polygons (MCP) (Mohr 1947) were calculated for each individual using the Adehabitat Package Version 1.4 (Calenge et al. 2006) in R 2.3.1 (R Development Core Team 2006) and the proportion of radio locations and MCP in different habitat types determined using the GIS.

Within the row crop site we defined 6 habitat types; winter wheat, fallow, wetlands, corn stubble, tilled land, and field edges. For the mixed use site we identified 5 habitat types; winter wheat, fallow, wetlands, mowed fallow, and grazed pasture. We used compositional analysis (Aebischer et al. 1993), based upon radio locations and MCP, to evaluate habitat preferences. The compositional analysis was performed using BYCOMP.SAS (Ott and Hoovey 1997) and, to obtain sufficient sample size, we combined data from both sites and aggregated habitat types into 5 categories; winter wheat, fallow, wetlands, edge, and short herbaceous (corn stubble, tilled land, mowed fallow, and grazed pasture). Additionally, survival was estimated using Kaplan-Meier staggered entry design (Kaplan and Meier 1958, Pollock et al. 1989). Standard errors were used to determine statistically significance differences in survival and mean home range size.

Results

The mean 100% MCP from the row crop site was larger (19.0 ha, SE = 10.4 ha) than that from the mixed use site (15.9 ha, SE = 7.3 ha), although differences were not significant due to high variance. Survival ($\hat{s}$ = 0.73, SE = 0.19) was higher in the mixed use site over 20 weeks compared to the row crop site ($\hat{s}$ = 0.33, SE = 0.19) over 15 weeks (Figure 2). Mortality of the radio-tagged birds from both sites was attributed mainly to predation (91%).

At the row crop site winter wheat, wetlands, and field edges were used less, and corn stubble more, than their availability based upon both the mean proportions of MCP and radio locations within those habitat types (Figure 3). In tilled land the mean proportion of MCP and radio locations indicate approximately equal use in relation to availability, while in fallow, based on the mean MCP use was equal to availability, but considerably higher than its availability based upon the mean proportion of radio locations (Figure 3). As at the row crop site about half of the area of the mixed use site was in winter wheat, which was utilized less than its availability (Figure 3). Fallow, mowed fallow, and wetlands were all used more than their availability, while based upon the mean proportion of MCP, grazed pasture was used equal to its availability, and less than its availability based upon the mean proportion of radio locations (Figure 3).

The compositional analysis using the aggregated data from both sites, and based upon MCP, ranked short herbaceous habitat as the most utilized habitat in relation to availability, with fallow, winter wheat, and edge ranked equally as second, followed by wetlands (Table 1a). No habitats were used significantly more than others (P = 0.05) but fallow and short herbaceous habitat were preferred over wheat, fallow over wetlands, short herbaceous over fallow, and wetlands over short herbaceous (Table 1a).

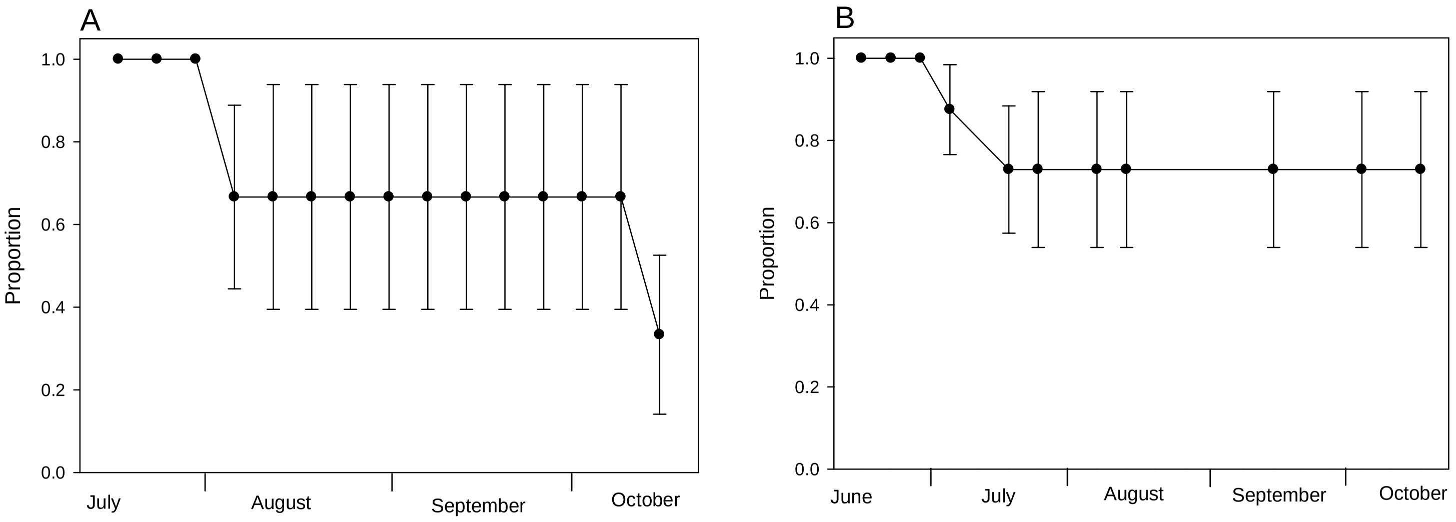

Figure 2: Estimated survival and standard error for spotted tinamous in the A) site dominated by row crops ($n = 3$) and B) the mixed grazing and row crop site ($n = 8$)

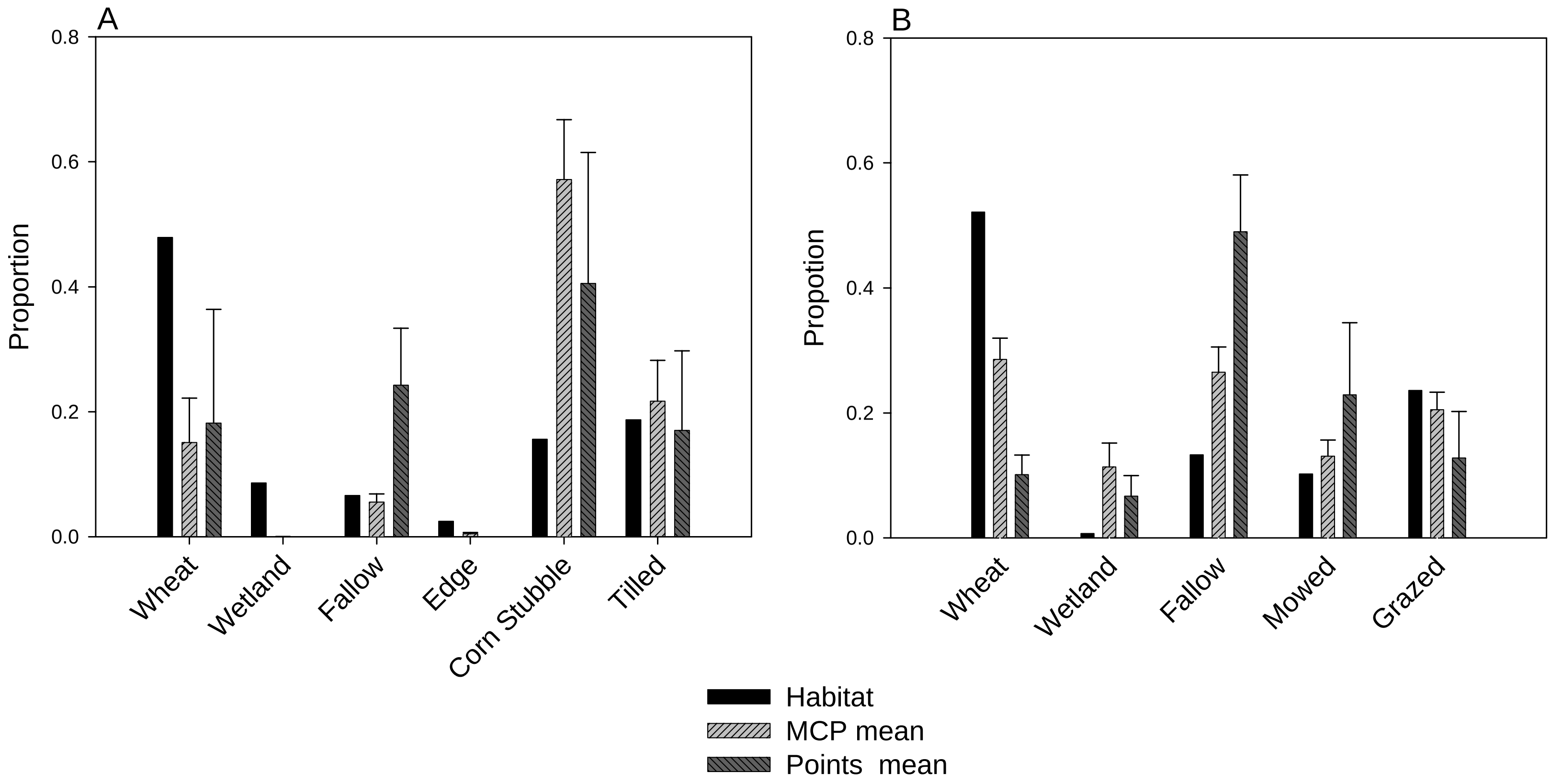

Figure 3: Proportional habitat use by spotted tinamous based on mean area of minimum convex polygon (MCP) and mean number of radio locations (Points) in relation to proportional availability of habitat types for A) the row crop site and B) the mixed use site. Error bars represent standard error.

Table 1: Results of compositional analysis based on a) minimum convex polygon (MCP) home ranges and b) radio locations. Higher ranking indicates greater use compared to availability. Within the matrix, (+) signifies that the row habitat is preferred over the column habitat, whereas a (-) signifies the opposite. Significant difference between habitats ($P < 0.05$) is indicated by (+++) or (−−−).

a

Habitat	wheat	wetlands	edge	fallow	short herbaceous	rank
wheat	.	.	.	−	−	2
wetlands	.	.	.	−	+	1
edge	.	.	.	.	.	2
fallow	+	+	.	.	−	2
short herbaceous	+	-	.	+	-	3

b

Habitat	wheat	wetlands	edge	fallow	short herbaceous	rank
wheat	.	−	.	−−−	−	0
wetlands	+	.	.	−	−	1
edge	.	.	.	.	.	0
fallow	+++	+	.	.	+	3
short herbaceous	+	+	.	−	.	2

Note: Because of low or no use a P-value for edge habitat could not be computed.

The same analysis using radio locations ranked fallow and short herbaceous habitats as the first and second most utilized habitats, respectively, in relation to availability, followed by wetlands (Table 1b). Winter wheat and edge were equally the least used in relation to availability (Table 1b). Fallow was utilized significantly more than wheat ($P = 0.05$) while fallow, short herbaceous, and wetlands were preferred over wheat, fallow and short herbaceous were preferred over wetlands, and fallow over short herbaceous (Table 1b).

Discussion

The mean range size of spotted tinamous at both sites was affected by movements related to changing habitat amounts and characteristics and cattle disturbance. At the row crop site as winter wheat reached ∼10cm in height birds began to utilize those areas, often exclusively and as the wheat matured to ∼25cm in height those areas were abandoned for areas with shorter vegetation. Within the mixed use site the largest movements by birds were related to disturbance by cattle.

The lower surival in the row crop dominated site is consistent with observations of Pinheiro and López (1999) who found lower abundances of spotted tinamous in agricultural land in southern Brazil compared to natural grasslands. Additionally, for the Galliformes there are multiple cases where increased intensification in land use has led to lower survival and declining populations (e.g. Berner 1988, Hill and Robertson 1988, Jansen et al. 2000, Malan and Benn 1999, Potts 1986). Based upon this, the observed differences in survival between the two sites are expected if the spotted tinamou is viewed as an ecological equivalent to the Galliformes. Admittedly, sample sizes are small, particularly for the row crop site; however, the rarity of spotted tinamous at the row crop site in 2003 and their absence from the site in 2004 suggest a real process rather than a statistical artefact.

Habitat preferences by the spotted tinamou, and the closely related Darwin's Tinamou (*Nothura darwinii*), within both natural and agricultural habitats, favour areas with relatively low (10-30 cm) and sparse vegetation (J.J.Thompson pers. obs., Bump and Bump 1969, Isacch and Martinez 2001, Leveau and Leveau 2004, Mosa 2004) and explains the pattern of habitat use at both sites. For example, use of winter wheat was most frequent when plants were 10-25 cm tall. Although wheat was generally avoided once it reached >25cm in height, birds then used it as escape cover.

The most preferred habitats; fallow, mowed fallow, and corn stubble all shared in common a well developed ground cover of herbaceous vegetation, both living and dead, that was not in excess of 50 cm and with little or no emergent vegetation above that level. Tilled land was used more as it was colonized by herbaceous vegetation, particularly clover (*Trifolium* spp.), and vegetative cover increased.

Spotted tinamous are often common in pastureland (J.J.Thompson pers. obs., Bump and Bump 1969, Menegheti 1985, Pinheiro and López 1999), as are Darwin's tinamou (J.J.Thompson pers. obs., Bump and Bump 1969, Mosa 2004), due to the low vegetative structure that is maintained through moderate grazing. At the mixed use site, however, pastureland was overgrazed so that ground vegetation was cropped near to ground level, which explains a lower than expected preference for grazed areas. The preference for relatively short vegetation also explains the avoidance of field edges in the row crop site. Field edges consisted of tall (>1m) and dense grass and also contained woody vegetation, which were avoided by the birds.

The difference in the use of wetlands among the sites appeared to be a function of the water levels within wetlands at each site. At the row crop site wetlands contained water and were avoided, where as at the mixed use site, wetlands were dry and contained suitable herbaceous cover along their perimeter that was utilized by the birds. It should be noted that although wetlands were not used by individuals at the row crop site, much of the fallow areas were not put into production due to their proximity to wetlands, subsequently wetlands were indirectly responsible for the availability of preferred habitats.

The preferences and differences in habitat use within and between sites are consistent with the results of the compositional analysis since fallow areas and the habitats comprising the short herbaceous category, while more variable, are the habitats most similar in structure to natural grasslands. Similarly, the quality of wetlands varies annually dependent upon precipitation, reducing interannual use, while row crop fields and edge were avoided or used considerably less in relation to their availability.

The preferences in habitat, size of home ranges, and survival that we observed were consistent with our expectations based upon existing knowledge of tinamou ecology, and the response of Galliformes and other bird species to the intensification of agricultural land use (Thompson 2004). From this study and others (Canavelli et al. 2003, Bellis et al. 2004, Demaría et al. 2002, Fernandez et al. 2003) it is apparent that the intensification of agriculture that has occurred in Argentina has resulted in similar negative ecological effects as observed in other regions.

The continued expansion and intensification of agriculture in Argentina suggests that pampean agroecosystems will continue to be degraded, with the most ecologically valuable systems being maintained in areas only suitable for extensive livestock production. Moreover, within intensively managed systems, fallow and areas unsuitable for production (i.e. wetlands) will increasingly become critical for biodiversity conservation.

Acknowledgments

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Gamebird Genetics

Release of Pen-reared Bobwhites: Potential Consequences to the Genetic Integrity of Resident Wild Populations

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In response to low encounter rates with wild northern bobwhite (*Colinus virginianus*; hereafter bobwhite) during bird dog field trials at Ames Plantation in Tennessee, a large-scale release program of pen-reared bobwhites was implemented in fall 2002. To evaluate genetic effects of pen-reared releases on wild populations, we monitored survival of pen-reared and wild bobwhites from fall release of pen-reared bobwhites through the breeding season and collected feather samples from wild, pen-reared, and free-ranging juvenile bobwhites following the first breeding season after the initial release. We used genotypes from 6 polymorphic microsatellite loci to measure genetic diversity and conduct population assignment tests. Wild bobwhites experienced greater fall-spring and annual survival than pen-reared bobwhites; however, pen-reared bobwhites experienced greater fall-spring and annual survival than reported in most other studies. Genetic diversity, number of alleles, and allelic richness were greatest in the wild, intermediate in the F1 generation, and lowest in the pen-reared populations. Likelihood analysis and cluster analysis indicated 20.4% and 33.6%, respectively, of juveniles captured after the first breeding season following release were ambiguous in population assignment; suggesting successful reproduction between wild and pen-reared individuals. These results suggest that large-scale releases of pen-reared bobwhite may result in negative impacts on genetic integrity of resident wild populations.

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Introduction

As northern bobwhite (*Colinus virginianus*; hereafter bobwhite) populations continue to decline throughout most of their range, the use of commercially produced bobwhites in lieu of wild bobwhites for bird dog training and field trials has become increasingly common (Hurst et al. 1993, Kozicky 1993, DeVos and Speake 1995, Dailey 2002). Pen-reared bobwhites are also used to augment existing wild populations to meet harvest demands far greater than sustainable by wild populations. Despite previous research (Baumgartner 1944, Buechner 1950, Sexson and Norman 1972) documenting the inability of pen-reared bobwhites to contribute substantially (i.e., survive and reproduce successfully) to wild populations, the release of pen-reared bobwhites remains a common practice used to meet short-term population goals. Whereas most releases of pen-reared bobwhites by private landowners have been for recreational purposes (i.e., shooting, dog training) and consist of relatively few birds; large-scale releases consisting of up to several thousand birds are becoming commonplace, especially at shooting preserves and field trial areas (Kozicky 1993, Sisson et al. 2000).

Large-scale releases of pen-reared bobwhites present potential for unforeseen ecological consequences. Disease transmission, displacement of wild bobwhites, and increased mortality of wild bobwhites due to numerical or functional predator

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responses to pen-reared bird releases may pose short to intermediate term risks (Hurst et al. 1993, Sisson et al. 2000, Hutchins and Hernandez 2003). However, longer term, less easily recognizable risks such as reduction in genetic variability of resident populations of wild bobwhites or introgression of maladaptive alleles is less well understood (Sexson and Norman 1972, Landers et al. 1991, DeVos and Speake 1995, Sisson et al. 2000, Hutchins and Hernandez 2003). Gutierrez (1993) suggested that if wild bobwhites exist in isolation at low densities and have adapted to local environmental conditions, large-scale release of pen-reared individuals may be detrimental to the genetic integrity of the population through dilution of locally adapted genepools. As such, a concern among most land managers and researchers is the likelihood of decreased natural genetic variability of wild populations or introgression of maladaptive genes following pen-reared release efforts (Wooten 1991, Hurst et al. 1993, Nedbal et al. 1997). However, these concerns are currently unsubstantiated because no research has investigated the effects of pen-reared bobwhite releases on the genetic structure of wild resident populations of bobwhites.

Ellsworth et al. (1988) reported less genetic variation in pen-reared than wild bobwhites. Breeding in captivity can produce extremely skewed reproduction and unintended selection which may reduce genetic variability (Roseberry et al. 1987, Ellsworth et al. 1988, Kozicky 1993) and facilitate the inadvertent selection of traits that may be maladaptive in the wild. Crossing of pen-reared with wild bobwhites has been suggested as a means to mediate loss of genetic diversity; however, backcrossing is *prima facie* evidence acknowledging genetic differentiation and directional selection in pen-reared populations.

Transference of pen-reared genes to wild populations necessitates that pen-reared bobwhites develop pair bonds, copulate, and successfully produce viable offspring with wild bobwhites. DeVos and Speake (1995) reported pen-reared bobwhites integrated into 72% of resident wild coveys; however, observations of pair bonds and reproduction of pen-reared and wild bobwhites was sparse. Confirmatory, genetic-based information of pen-reared and wild bobwhite production is non-existent. Secondly, pen-reared bobwhites must survive until the breeding season. Given the relatively low survival of pen-reared bobwhites (Fies et al. 2000, Oakley et al. 2002, Perez et al. 2002), releases conducted during the fall may not pose a threat to native gene pools because, in most instances, pen-reared bobwhites do not survive to the breeding season and thus do not participate in reproduction. However, Frye (1942) reported up to 58% fall-spring survival for pen-reared bobwhites released in Florida. Given this fall-spring survival, pen-reared bobwhites released in the fall may survive to the breeding season, compete for mates, and subsequently reproduce with wild bobwhites; thereby contributing to local gene pools.

Our objectives were to estimate fall-spring and annual survival of pen-reared and resident wild bobwhites at Ames Plantation in southwest Tennessee and to evaluate the genetic consequences of pen-reared bobwhite releases on the genetic structure of the local wild bobwhite population during the first breeding season following initiation of a large-scale release program. We hypothesized that the release of pen-reared bobwhites would result in the introgression of pen-reared alleles in the F1 generation.

Study Area

Our study was conducted at Ames Plantation in Hardeman and Fayette counties, Tennessee (89° 11′ W, 35° 8′ N). Owned and operated by the Hobart Ames Foundation, Ames Plantation is home to the National Bird Dog Championship and also serves as a branch of the University of Tennessee Agricultural Experiment Station system. Of the 7,552 ha plantation, approximately 2,429 ha were used to host field trials and was managed intensively for wild bobwhites. Land cover on the field trial courses consisted predominantly of corn (*Zea mays*) and soybean (*Glycine max*) row crop fields interspersed with idle and perennial grass fields and woodlands. Prescribed burning, disking, rotational agriculture, and selective herbicide applications were used to maintain early succession plant communities within open lands and pine woodlands. Sorghum (*Sorghum vulgare*), soybean, and wheat (*Triticum aestivum*) food plots were planted in small (<1 ha) patches. For a more complete study area description, see Seckinger (2004).

Despite the success of habitat management efforts to elevate and maintain relatively high densities of wild bobwhites; encounter rates with bobwhite during field trials still remained below desired levels. Consequently, Ames Plantation instituted a pen-reared bobwhite release program in fall 2002 to elevate bobwhite densities to desired levels (1 bird/0.5 ha) for conducting field trials. Approximately 3,200 pen-reared bobwhites were released each fall (1 October) from 2002-2004.

Methods

Capture, Marking, And Releasing

Pen-reared bobwhites were purchased from a commercial producer (Clear Creek Farms, Lamar, Mississippi, USA) and held on site for 95-105 days prior to release in 2 holding pens. Each holding pen consisted of a 4.6-m 6.1-m enclosed brooding area with a 3.7-m × 6.1-m × 45.7-m flight pen. Commercial feed (28% crude protein, medicated with BMD and a cocidiostat) and water were provided *ad libitum*.

Prior to release (4-14 days), we sexed, weighed, banded with a #8 aluminum leg band, and fitted a 5-6 g pendant style radio transmitter (American Wildlife Enterprises, Tallahassee, Florida, USA) to a sample (2002, n = 191; 2003, n = 216) of these pen-reared bobwhites. On the evening prior to the release (1 October each year), 1-2 radiomarked birds were placed into each of 160 release boxes containing 18-19 non-radiomarked pen-reared bobwhites. All bobwhites within each box were then released the following morning at 1 of 160 release sites distributed over the study area. We released 3,200 pen-reared bobwhites each fall, with an additional 200 pen-reared bobwhites released during January 2003. Release sites were selected to provide cover in close proximity to food resources with most release sites situated in dense food plots of sorghum or corn or a natural herbaceous community. Food (7.6 L of sorghum) and water (1.9 L) dispensers were located at each of the release sites.

Wild resident bobwhites were captured during the fall and winter of each year from 2000-2004 with baited walk-in funnel traps (Stoddard 1931) or by night netting (Truitt and Dailey 2000). We also captured periodically additional bobwhites during the breeding season using call-back traps and by night-netting. Captured wild bobwhites were identified and radiomarked in a similar fashion as the pen-reared bobwhites, except wild bobwhites were released at the capture site immediately after radiomarking. Capture, handling, tagging, and radiomarking procedures were consistent with the American Ornithologist's Union Report of Committee on the Use of Wild Birds in Research (American Ornithologists' Union 1988).

We used a programmable scanning receiver with a 3-element Yagi antennae to monitor radiomarked pen-reared and wild bobwhites $\geq$5 days/week from 1 October 2002-30 September 2004. Radio transmitters operated on 148.000-151.000 MHz wavelengths and were equipped with a 12-hr motion sensitive mortality switch. When a mortality signal was detected, we located the transmitter and determined fate of the radiomarked bird using evidence at the recovery site (i.e., bird remains, scat, tracks, whitewash) and transmitter damage (Dumke and Pils 1973). Intact birds for which no apparent cause of mortality could be determined readily were considered to have died due to exposure.

Survival Analysis

We used Cox's partial likelihood regression (Cox 1975) in PROC PHREG (Allison 1995) to estimate survival and test hypotheses of no difference in proportional hazard between pen-reared and wild bobwhites and sex. We calculated survival for 2 post-release time intervals (fall-spring, 183 days; annual, 365 days) for each year (2002-2003 and 2003-2004) beginning on the release date of pen-reared bobwhites (1 October). Wild bobwhites radiomarked prior to the release of pen-reared bobwhites entered the survival analyses on the release date of the pen-reared bobwhites for each year (i.e., survival estimates of wild bobwhites began on the same day as pen-reared bobwhites). We right-censored birds due to transmitter failure, suspected emigration from the study site, or trap-related mortality on the last date a signal was recorded. Right-censoring accounts for incomplete data that is not a result of a failure to survive during the study period and is therefore "censored" during analysis (Martinussen and Sheike 2006). Wild bobwhites that were marked in one year and survived to the next were right-censored on 30 September and introduced as new independent observations on 1 October. Pen-reared bobwhites surviving >365 days (n = 5) were not included in the subsequent year's estimate because we desired to measure only post-release survival of pen-reared bobwhites up to 1 year. We assumed sexes were sampled randomly, individual survival times were independent, the censoring mechanism was random, and capturing, handling, and radiomarking did not affect survival (Pollock et al. 1989). Results were considered significant at $\alpha = 0.05$. Because variation in annual survival of bobwhites has been well documented (Rosene 1969, Burger et al. 1995), we analyzed each year independently and did not test for year effects.

Genetic Analysis

Feather Samples.- We collected feather samples from wild, pen-reared, and F1 generation bobwhites during both years of study; however, because funding for the genetic analyses was limited, we chose only to analyze the 2002-2003 feather samples because pen-reared bobwhite survival was greatest for this time interval and would likely represent the "worst case" scenario of pen-reared bobwhite contribution to production. We collected 5-10 body feathers from the ventral tract of each of approximately 200 wild bobwhites captured from January-August 2002, 900 randomly selected pen-reared bobwhites released in the fall 2002, and from all pen-reared bobwhites released in January 2003. From September 2003 to May 2004 we captured and collected feather samples from approximately 200 juvenile bobwhites (F1 generation) from multiple coveys within the study area using baited walk-in funnel traps. To avoid cross-contamination, feather samples from each individual were stored separately in dry envelopes. Bird handling and feather sampling were conducted under the auspices of the Mississippi State University Institutional Animal Care and Use Committee (permit #01-051).

We selected randomly 50 feather samples from each of the wild, pen-reared, and F1 generation groups. DNA was extracted from feather tips using a Qiagen DNeasy Tissue Extraction Kit (Qiagen Inc., Valencia, Ca) combined with Dithiothreitol (DTT) to aid in the breakdown of the keratinized feather shaft. Six di- and tetra-nucleotide microsatellite markers (K. W. Fok, University of Georgia, unpublished data, Fok and Parkin 2003, Schable et al. 2004) were amplified in 10 μl polymerase chain reactions (PCR) containing DNA template, Takara Ex-Taq DNA polymerase, 10X PCR buffer (containing 20 mM Mg^{+2}), 2.5 mM each dNTP (pH 7$\sim$9), and 1 μM each fluorescent-labeled primer (Proligo LLC, Boulder, Co). PCR reactions were conducted with an initial denaturation of 5 min at 95°C, followed by 40 cycles of 95° C for 30 sec, 30 sec at the locus-specific annealing temperature (Table 1), and 72° C for 30 sec. Cycling was followed by a final extension period of 20 min at 72° C. Following amplification, products were identified and sized by capillary electrophoresis on a DNA Sequencer (CEQ 8000XL, Beckman-Coulter Inc., Fullerton, Ca). Fragments representing pairs of alleles at each locus (i.e., genotypes) were generated for each individual in a population and binning analysis of alleles at each locus was conducted to ensure accurate scoring of fragment sizes and alleles.

Genetic Differentiation.- Deviations from Hardy-Weinberg (HW) and linkage equilibrium were calculated using Program GENEPOP 3.3 (Raymond and Rousset 1995). To reduce the probability of Type I er-

Table 1: Locus identity, annealing temperatures (°C), and accession numbers for each locus used to examine introgression of pen-reared individuals with wild bobwhite populations on Ames Plantation, Tennessee, 2002−2003.

Locus	Annealing Temp.	Accession No.
LEI-142	68	X83257
LEI-160	66	X85523
LEI-70	63	X82869
LEI-197	63	Z83776
NBGP-9	57	AY522966
NBGP-11	57	AY522968

ror due to multiple testing, we used sequential Bonferroni to adjust nominal significance levels (Rice 1989). Allele frequencies, gene diversity, number of alleles, allelic richness, and inbreeding coefficients (F_{IS}) were calculated for each population (wild, pen-reared, F1 generation) using Program FSTAT 2.9.3 (Goudet 2001).

Degree of introgression of pen-reared and wild individuals was analyzed using assignment test procedures of Paetkau et al. (1995). Individuals were first assigned to likely source populations using Program WHICHRUN 4.1 (Banks and Eichert 2000), designating the pen-reared bobwhite group as the critical population (stringency = 2). A second analysis was conducted designating the wild bobwhite group as the critical population (stringency = 2). Likelihood values were calculated for each individual and the $\log_{10}$ of the quotient of the critical population's likelihood divided by the most likely population's likelihood was calculated to generate a LOD score. Individuals possessing LOD values greater than stringency values belonged to the defined critical population. Most likely population probabilities were also calculated and the probability (P) an individual belonging to the most likely (ML 1) population divided by the probability of the individual belonging to the second most likely population (ML 2) was calculated. Values <3.00 were characterized as ambiguous in population assignment.

Bayesian analysis of allele frequencies was conducted to evaluate admixture in the F1 generation using Program STRUCTURE 2.0 (Pritchard et al. 2000). Posterior probabilities of K (number of populations) were used to assign individuals to populations and using a prior population model (K = 3, Burnin = 10,000, MCMC Reps = 10,000) and correlated allele frequencies.

Results

Survival

We used 409 pen-reared and 316 wild bobwhites to estimate survival. We right-censored 10 pen-reared bobwhites due to suspected emigration from the study site, 4 due to transmitter failure or transmitter related mortality, and 5 due to trap related mortality. We right-censored 12 wild bobwhites due to suspected emigration, 27 to transmitter failure or transmitter related mortality, and 4 to trap related mortality.

Fall-spring survival (183 day) did not differ between sexes in 2002 ($\chi 1^2$ = 1.09, P = 0.296) or 2003 ($\chi 1^2$ = 0.03, P = 0.873). Wild bobwhites experienced greater fall-spring survival than pen-reared bobwhites in 2002 ($\chi 1^2$ = 3.98, P = 0.046) and 2003 ($\chi 1^2$ = 8.82, P = 0.003; Table 2). Annual survival was similar between sexes in 2002 ($\chi 1^2$ = 0.02, P = 0.882) and 2003 ($\chi 1^2$ = 1.56, P = 0.211). Wild bobwhites had greater annual survival in 2002 ($\chi 1^2$ = 5.83, P = 0.016) and 2003 ($\chi 1^2$ = 17.90, $P < 0.001$; Table 2).

Table 2: Survival (S) of pen-reared and wild northern bobwhite for 183 and 365 days following release (1 October) of pen-reared northern bobwhite at Ames Plantation, Tennessee, 2002−2004.

		Pen-reared			Wild			
Period	Year	*n*	S	SE	*n*	S	SE	P-value
183-days	2002−2003	190	29.8	0.03	124	44.6	0.05	0.046
	2003−2004	219	12.2	0.02	137	32.9	0.05	0.003
365-days	2002−2003	190	3.2	0.01	150	8.4	0.02	0.016
	2003−2004	219	0.5	0.00	166	4.5	0.01	<0.001

Genetics

Four individuals from the wild, 3 from the pen-reared, and 1 from the F1 generation groups were removed prior to analysis due to missing data at 3 or more loci. We found no evidence of HW or linkage disequilibrium; indicating a random union of gametes and independence of loci within each group (wild, pen-reared, F1 generation).

Wild and pen-reared birds shared 44 common alleles across all loci (Figure 1). Allele frequency analysis indicated 4 instances where alleles were specific to pen-reared and F1 generation populations but not found in the wild population (Locus LEI-97 [133, 153, 155], Locus LEI-142 [128]; Figure 1). There were 14 instances of private alleles: 11 specific to wild birds, 2 specific to pen-reared birds, and 1 specific to F1 generation birds (Figure 1). However, only 2 alleles from the wild population (LEI 142 [106], NBGP 9 [194]) exceeded the threshold frequency of 0.05 required to ensure that the alleles are a product of population differences and not random sampling (Beaumont et al. 2001).

Gene diversity, number of alleles, and allelic richness averaged across all loci were greatest in the wild population, intermediate in the F1 generation population, and lowest in the pen-reared population (Table 3). Overall, genetic diversity estimates were high for all three populations (range = 0.790-0.841; Table 3). Relative to the wild population, the F1 generation population exhibited less genetic diversity, possibly due to the introduction of pen-reared birds (Table 3).

Likelihood ratio analysis of the 49 F1 generation birds indicated that 30 individuals (61.2%) were most likely sired from two pen-reared adults ($P_{(ML1/ML2)} > 3.00$), and 13 of those 30 individuals were assigned to the pen-reared population when LOD values were compared to a stringency value of 2 (<1/100 chance of error). Nine individuals (18.4%) most likely were sired from two wild adults ($P_{(ML1/ML2)} > 3.00$), but only 1 individual was significantly assigned to the wild population when LOD values were compared to a stringency value of 2. Ten individuals (20.4%) were ambiguous in population assignment ($P_{(ML1/ML2)} < 3.00$; Figure 2). This ambiguity may reflect possible hybrid offspring that resulted from the cross of wild and pen-reared adults.

Estimation of the proportion of membership of individuals into clusters was successful for individuals in the wild and pen-reared population. Cluster 1 grouped wild individuals with a high proportion of membership (q1 = 0.985) whereas cluster 2 grouped the pen-reared individuals with a high proportion of membership (q2 = 0.980). However, members of the F1 generation population were derived from the wild population cluster (q1 = 0.311), the pen-reared population cluster (q2 = 0.353), and from its own F1 generation cluster (q3 = 0.336); suggesting that 33.6% of individuals in the F1 generation cluster were possible hybrids that could not be placed into either the wild or pen-reared populations due to an

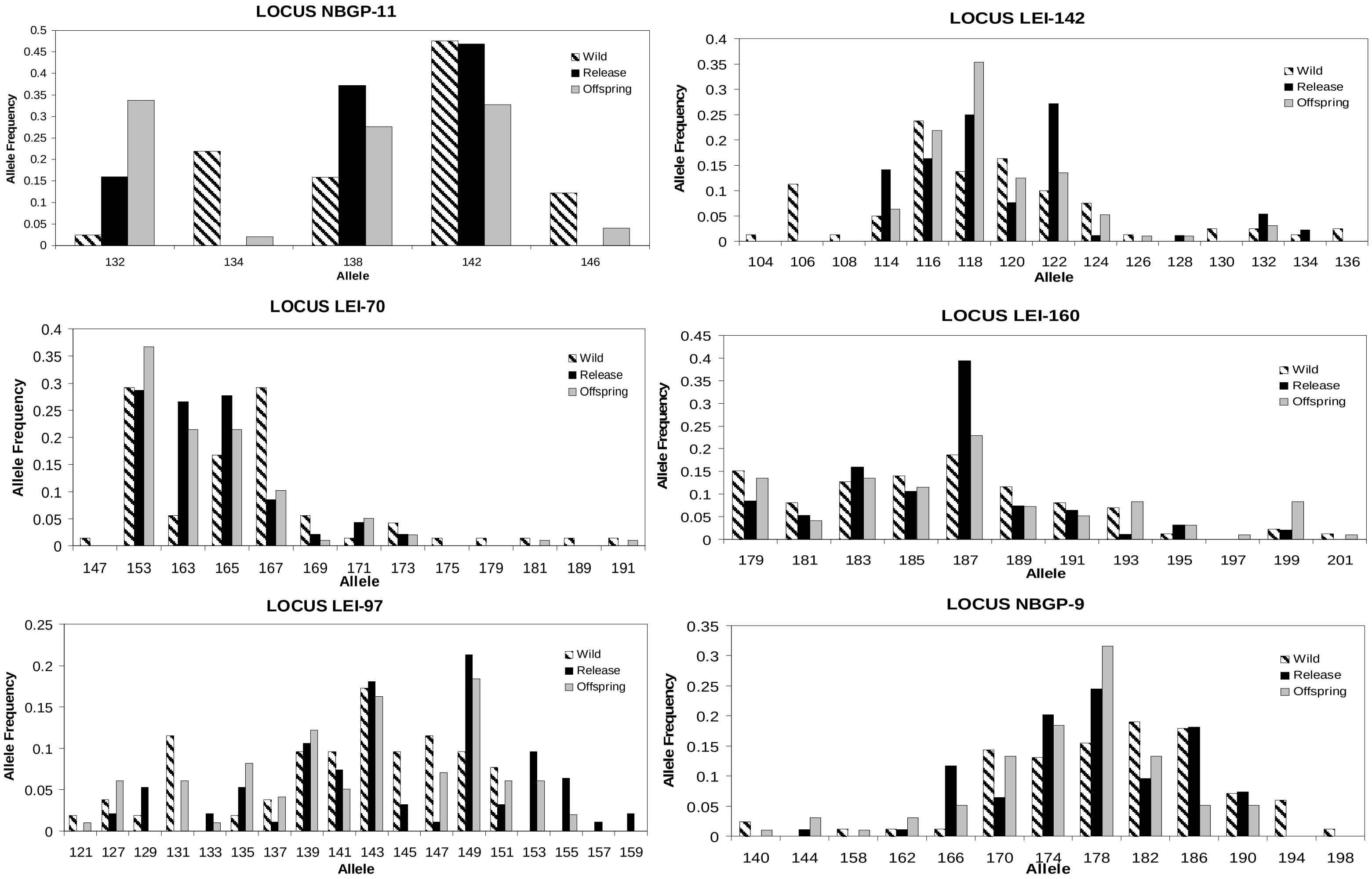

Figure 1: Allele frequencies per locus per population of wild, pen-reared, and F1 generation bobwhite at Ames Plantation, 2002.

Table 3: Gene diversity (H), number of alleles (N), allelic richness (RS), and inbreeding coefficient (FIS) averaged over all loci for wild, pen-reared, and F1 generation bobwhites at Ames Plantation, Tennessee in 2002.

Population	H	N	RS	FIS
Wild	0.841	11.300	10.250	0.035
Pen-reared	0.790	9.000	8.116	-0.029
F1 Offspring	0.814	10.000	8.920	0.032

admixture of alleles.

Discussion

Our survival results were consistent with those of other studies (DeVos and Speake 1995, Fies et al. 2000, Perez et al. 2002) in that wild bobwhites experienced greater survival than pen-reared bobwhites. However, survival of pen-reared bobwhites in our study (12.2-29.8% fall-spring, 0.5-3.2% annual) was substantially greater than survival reported in most other studies of pen-reared bobwhites, except for Frye (1942). Oakley et al. (2002) reported 0-11% fall-spring survival for pen-reared bobwhites in Maryland whereas Roseberry et al. (1987) attained only 15% recovery of pen-reared bobwhites in Illinois. DeVos and Speake (1995) reported winter-spring (154-day interval) survival of approximately 18% for pen-reared bobwhites in Alabama. All game-farm birds in Fies et al. (2000) and Perez et al. (2002) died within 3 and 12 weeks, respectively, of release. Fall-spring survival of wild bobwhites in our study were similar to that on intensively managed plantations in Georgia (47.2%, 10-48.2%; Burger et al. 1998, Sisson et al. 2000, respectively) but greater than that reported for un-managed farmlands in Missouri (15.9%, Burger et al. 1995).

Guthery and Lusk (2004) suggested inherent negative bias in bobwhite survival from telemetry studies due to effects of radiomarking. However, Corteville (1998) reported similar survival for wild bobwhites fitted with mock transmitters as those with leg bands only. Although no studies of transmitter effects on released pen-reared bobwhites have been conducted, we assumed that if transmitters negatively biased survival in wild bobwhites, similar biases would occur with pen-reared bobwhites. Secondly, pen-reared bobwhites were radiomarked and then released without an "acclimation" period whereas some wild bobwhites were radiomarked prior to the monitoring period and thus had greater time to adjust to radiomarking. Insofar as the above sources of bias may have influenced survival, our survival estimates likely reflect the lower bounds of pen-reared bobwhite survival. Regardless of potential telemetry induced bias, survival of pen-reared birds in our study was substantially greater than that reported in most other radio-telemetry studies; with several pen-reared bobwhites surviving to the breeding season.

Hybridization of genetic stocks has often been associated with beneficial results such as increased genetic diversity (Roy et al. 1994, Randi and Bernard-Laurent 1999) and greater survival and seasonal production (Niewoonder et al. 1998). However, several instances have been reported where purposeful or incidental re-stocking of species has led to hybridization, introgression of captive alleles, and eventual detrimental effects on native populations (Templeton 1986, Rhymer and Simberloff 1996).

As such, a common, although previously unsubstantiated, concern with bobwhite release programs is that pen-reared individuals may not be adapted to the local environment and may hybridize with wild individuals; thereby decreasing overall fitness of the local resident population (Rhymer and Sim-

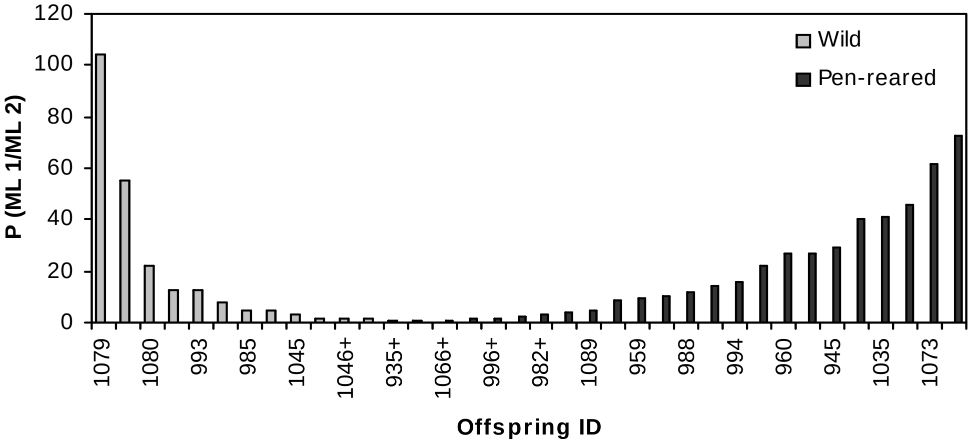

Figure 2: Probability of F1 generation bobwhites belonging to the most likely population (ML 1) divided by the probability of belonging to the second most likely population (ML 2) at Ames Plantation, 2002. $^{+}$Individuals with a probability ratio approaching 1.00 represent ambiguous population assignments and are therefore probable hybrids. Excludes individuals that significantly belonged to the critical population.

berloff 1996). Nedbal et al. (1997) reported that wild bobwhites originating from south Texas did not contribute to reproduction when transplanted to east Texas due to differences between subspecies. Pen-reared and wild individuals shared several alleles across all 6 loci. However, we observed greater than five times as many private alleles in the wild population than in the pen-reared population (although only 2 possessed frequencies >0.05); suggesting that the wild population possessed greater genetic variability at these particular loci. Similarly, Ellsworth et al. (1988) reported lower percentage of polymorphic loci in game farm than wild bobwhites.

Of greater concern was the subsequent lack of genetic diversity, number of alleles, and allelic richness observed in the F1 generation relative to the wild population. Cross-breeding of pen-reared and wild individuals was likely responsible for the reduced genetic variability we observed in the F1 sample. Less genetic variability in pen-reared populations is plausible given that most captive breeding systems expose birds to artificial selective forces (Roseberry et al. 1987, Ellsworth et al. 1988, Kozicky 1993) and transferring low genetic variability would occur when pen-reared and wild individuals cross-breed.

Pen-reared individuals released during the fall of 2002 survived to and, as evidenced by our genetic analysis, reproduced successfully during the 2003 breeding season. Assignment tests demonstrated that pairs of pen-reared adults bred, pairs of wild adults bred, and some pen-reared adults may have bred with wild adults. We could not assign 20.4-33.6% of F1 generation individuals to either the wild or pen-reared population. Results from cluster analysis (33.6%) were greater than the estimate provided from the likelihood ratio analysis (20.4%). However, because the threshold value of 3.00 (by which $P_{(ML1/ML2)}$ was compared) was a user-defined value, it may have produced more stringent results when compared to cluster analysis. There is no specified value of $P_{(ML1/ML2)}$ to indicate a hybrid, only that as the value approaches 1.00 there is ambiguity in population assignment (Banks and Eichert 2000).

This ambiguity in population assignment for several F1 generation individuals was likely due to the high proportion of shared alleles between the wild and pen-reared individuals. However, our assignment test procedures utilized allele frequencies and not allele identity to classify individuals into populations. Future studies could use parentage analysis on a larger sample of individuals and loci to determine with greater accuracy if these potentially hybrid individuals were truly hybrids.

Differential capture probabilities between F1 generation and wild bobwhites may have influenced substantially our results. Although pen-reared bobwhites have a greater tendency for recapture after release than wild bobwhites (Roseberry et al. 1987, L. W. Burger, Mississippi State University, personal communication); biases in capture probability of pen-reared offspring and offspring of pen-reared and wild hybrids is unknown. Similarly, capture probability of wild or hybrid offspring coveys containing pen-reared individuals is unknown. If offspring of two pen-reared birds or hybrid offspring exhibit greater capture probability than wild bobwhites, our results likely overestimated the introgression of pen-reared genetic material into wild populations. Therefore, within the limitations of our study, we recommend that our results be considered only as a cursory examination of determining the effects of pen-reared bobwhites on the genetic structure of local wild populations.

Management Implications

Reduction of genetic variability in wild bobwhite populations has been a point of concern for several decades and only recently have researchers been able to feasibly study genetic structure and variability of wild populations. Observing that 73% of resident wild coveys contained pen-reared bobwhites on areas where pen-reared releases occurred, DeVos and Speake (1995) speculated that cross-breeding may produce biologically inferior offspring. However, no studies have yet examined the survival of F1 generation hybrids raised *in situ* by wild bobwhites. Given the 29.8% fall-spring survival of 3,200 pen-reared bobwhites released in 2002, a conservative estimate of 954 pen-reared bobwhites were alive at the beginning of the 2003 breeding season (1 April). Wild bobwhite density on the field trial course at Ames Plantation was estimated to be approximately 1 bird/0.6 ha (3,981 birds) during fall 2002 with an expected 2003 breeding population of 1,776 birds, assuming 44.6% survival. Pen-reared birds represented approximately 35% of the total 2003 breeding population. Therefore, we believe our estimates that 20.4-33.6% of the F1 generation birds captured during the fall of 2003 may have been pen-reared-wild hybrids are plausible; suggesting that cross-breeding of pen-reared and wild bobwhites likely occurred.

Given the relatively high fall-spring survival of pen-reared bobwhites combined with our observations of the genetic diversity of the F1 generation, we recommend that managers and researchers consider the potential effects of large-scale releases of pen-reared bobwhites on the genetic integrity of wild bobwhite populations. Additionally, we suggest that future research focus on genetic analysis of populations for multiple generations in areas where releases of pen-reared birds occur. Because we were only able to examine genetic variability for one generation following release of pen-reared bobwhites, we view this research as a precursor in determining the effects of pen-reared bobwhite releases on resident wild populations. We suggest future studies should incorporate a greater number of loci and larger sample sizes of individuals over multiple generations before definitive conclusions regarding the effects pen-reared bobwhite releases on the genetic variability of local wild bobwhite populations can be determined.

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Habitat

Effects of Sex, Age, and Habitat on Northern Bobwhite Spring Dispersal Patterns

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Information on northern bobwhite (*Colinus virginianus*) dispersal patterns is crucial for implementing effective management strategies. Researchers have examined bobwhite dispersal, but information on how habitat affects dispersal patterns is lacking. We examined the effects of habitat, sex, and age on bobwhite spring dispersal patterns in a southern Georgia agricultural landscape during 2002-2003. Of 101 birds used in our analyses, 29.7% (4.6 SE) dispersed an average of 1,835m (194 SE). We fit 9 logistic regression models to predict bobwhite dispersal probability. The selected best model (Akaike weight $[\omega]$ = 0.58) included age, proportions of closed-canopy pine within winter home ranges (CCPN), and an age*CCPN interaction term. Adults with higher proportions of closed-canopy pine within their winter home range were more likely to disperse (β = 0.18, 0.06 SE). Because of greater experience, adults may perceive habitat differently than juveniles, which could influence adult tendency to disperse. However, a significant portion of birds from both age classes will likely disperse every spring, regardless of habitat quality. Although dispersal may allow bobwhite populations to persist in fragmented landscapes, efforts to increase bobwhite populations at the local scale are hindered if emigration exceeds immigration. Therefore, it is important to consider landscape quality and management unit size when determining which areas are most likely to respond to management and the proper management strategy needed to achieve bobwhite population objectives.

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Key words: age, closed-canopy pine, *Colinus virginianus*, dispersal, Georgia, habitat, metapopulation, northern bobwhite, scale, sex

Introduction

Information on northern bobwhite dispersal patterns is crucial for implementing effective management strategies. Site-specific management will always be an important component of any effort to increase bobwhite populations, but management strategies aimed at increasing bobwhite populations on a regional scale must consider landscape level aspects of bobwhite ecology in order to be effective (Brady et al. 1993, Roseberry 1993, Burger 2002, Fies et al. 2002). Areas on the landscape with the greatest potential to respond positively to management need to be identified so that management efforts can be implemented in an efficient and effective manner (Roseberry and Sudkamp 1998, Schairer et al. 1999). Most of the landscapes in which bobwhites exist are highly fragmented; therefore, an understanding of the role of metapopulation processes in regulating regional populations is critical for management programs designed to increase bobwhite populations. The development of spatially explicit population models has been advocated (Burger 2002, Fies et al. 2002) and may be particularly valuable because these models allow managers to predict the possible effects of large-scale management strategies (Conroy et al. 1995, Dunning et al. 1995). All of these endeavors will require information on dispersal rates and distances and how factors such as sex, age and habitat affect these parameters. Dispersal information would also be useful for site-specific management. Knowledge of immigration/emigration ratios for management units and how these ratios may be affected by management unit size and the surrounding landscape will be useful in predicting population response to different management strategies. It

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would be particularly useful to know if and to what degree emigration would decrease as habitat quantity/quality changed on a management unit.

Several studies have examined bobwhite mobility/dispersal (e.g., Stoddard 1931, Duck 1943, Loveless 1958, Kabat and Thompson 1963, Urban 1972, Smith et al. 1982, Fies et al. 2002, Townsend et al. 2003, Terhune et al. 2006). However, most of the early efforts relied on recoveries of leg-banded individuals. Large sample sizes are required to estimate dispersal distances using these techniques (Paradis et al. 1999). Also, these studies likely underestimated dispersal distances because detection probabilities (recovery rates) generally decline with increasing dispersal distance (Koenig et al. 1996). Radio-telemetry provides an opportunity to more accurately detect dispersal and estimate dispersal distances. However, many telemetry studies also likely underestimate dispersal distances because birds that leave the study area are often censored from analysis. Recent studies of bobwhite dispersal (Fies et al. 2002, Townsend et al. 2003) have produced more accurate estimates of dispersal distances and rates because they were conducted using radio-telemetry without study area boundary constraints. However, these studies did not examine the effect of habitat on dispersal patterns. The effects of habitat configuration and composition on bobwhite dispersal patterns may have profound management consequences.

We quantified bobwhite spring dispersal (rate and distance) and examined the effects of sex, age, and habitat on spring dispersal. Although bobwhites may move long distances during any time of the year, most dispersal events occur in the spring prior to the breeding season (Fies et al. 2002, Townsend et al. 2003, Folk 2006). We defined spring dispersal as a permanent movement from a winter range to a breeding range.

Study Area

We conducted this study on a 133 km^2 area in western Laurens County, Georgia. This fragmented landscape, typical of the modern southeastern USA, was comprised of row-crops (13%), pasture/hayfields (12%), closed canopy pine plantations (23%), hardwoods/mixed stands (33%), and early successional habitat (10%). Most crop fields were planted in cotton (*Gossypium hirsutum*), but some were planted in peanuts (*Archis hypogaea*), corn (*Zea mays*), or soybeans (*Glycine max*). A few fields were planted in winter wheat (*Triticum aestivum*) or rye (*Secale cereale*). Pastures and hayfields were dominated by bermudagrass (*Cynodon dactylon*) and/or bahiagrass (*Paspalum notatum*). Closed-canopy pine plantations were planted stands of either loblolly (*Pinus taeda*) or slash pine (*P. elliotii*) that had reached canopy closure and had little to no vegetation in the understory. Areas classified as early successional were dominated by weeds and/or short brush. These included abandoned fields, herbaceous strip-cover that either bordered or passed through the interior of crop fields (habitats provided by the Georgia Bobwhite Quail Initiative, Georgia Department of Natural Resources 1999), planted pines / clearcuts that had not reached canopy closure, hedgerows, and fencerows. We created a computerized (vector) habitat map of the study area by referencing U.S. Geological Survey 1993 Digital Orthophoto Quarter Quadrangles (DOQQ) in ArcView® (Environmental Systems Research Institute, Inc., Redlands, California). Although the photographs were 10-years old, we verified land cover types through infield ground truthing using Global Positioning System equipment.

Methods

We captured bobwhites during January 2002-April 2002 and November 2002-April 2003 using wire walk-in funnel traps (Stoddard 1931) baited with cracked corn. Captured bobwhites weighing >140g were banded, equipped with a 6.4-6.9 g pendant-style (necklace) radiotransmitter and released at the trap site. We determined sex and age (juvenile and adult) using techniques of Rosene (1969). All trapping, handling, and marking procedures were consistent with guidelines in the American Ornithologists' Union Report of Committee on

Table 1: Spring dispersal rates and distances for 101 radio-marked bobwhites monitored in Laurens County, Georgia during 2002-2003.

		Dispersal Rate (%)			Dispersal Distance (m)		
		n	Rate	SE	n	Mean	SE
Female	Juvenile	35	25.7	7.5	9	2,184	341
	Adult	15	26.7	11.8	4	2,150	790
	Pooled	50	26	6.3	13	2,173	319
Male	Juvenile	35	34.3	8.1	12	1,339	203
	Adult	16	31.3	12	5	2,146	575
	Pooled	51	33.3	6.7	17	1,576	230
Pooled	Juvenile	70	30	5.5	21	1,701	204
	Adult	31	29	8.3	9	2,148	442
	Pooled	101	29.7	4.6	30	1,835	194

the Use of Wild Birds in Research (American Ornithologists' Union 1988) and those of the University of Georgia, Institutional Animal Care and Use Committee (IACUC Protocol No. A2003-10109-0).

We located bobwhites 4-7 times per week during the breeding season (15 April-15 September) by homing (White and Garrott 1990) to within 25-50 m. Locations prior to the breeding season were obtained 3-5 times per week. When radio contact was lost, we systematically searched from vehicles for lost birds within 5 km of their last known location. We plotted locations onto aerial photos and later transferred to a Geographic Information System format using ArcView®.

We detected spring dispersal using the vectored dispersal detection technique (Kenward 2001) in RANGES V (Kenward and Hodder 1996). This technique tests if n new locations are outside the distribution of all previous N locations in a single direction. The detector begins by calculating the arithmetic mean center (Ac) of the first N = 3 locations and buffers this center by the upper confidence limit of distances of points to the Ac for a selected α level (we used α = 0.05). It then calculates the Ac of the next n = 3 locations and constructs a line (vector) through the two Ac points. If all of the orthogonal distances of the n points along the vector are outside of the confidence limit of the first N locations, then dispersal is flagged. If not, N incrementally increases by 1 and the routine begins again. We considered the first n location that was part of a set of n = 3 locations where dispersal was flagged to be the first date of dispersal. To determine when dispersal ended, we treated the last breeding season location as the first winter location (and *vice versa*) and considered the n that was part of a set of n = 3 where dispersal was flagged to be the last date of dispersal. Dispersal was not detected "in reverse" for a few dispersers. This was likely due to the fact that the distances of breeding season locations to their Ac were generally much greater than the distances of winter locations to their Ac. When this occurred, dispersal locations were considered to be part of the breeding season location set. Dispersal distance was determined by measuring the distance between the Ac of winter locations and the Ac of breeding season locations for each disperser.

We employed several additional decision rules

Table 2: Ranking of candidate models used to predict spring dispersal of radio-marked bobwhites monitored in Laurens County, Georgia during 2002-2003.

Model	K[a]	**Log Likelihood**	QAICc[b]	**Δ QAICc**	**Akaike Weight**
Age + CCPN[c] + Age*CCPN	4	-56.64	113.72	0.00	0.58
Null Model	1	-61.44	116.72	2.56	0.16
Sex	2	-61.12	117.75	4.03	0.08
Age	2	-61.44	118.35	4.63	0.06
Age + ERSC[d] + Age*ERSC	4	-59.62	119.27	5.55	0.04
CCPN + ERSC	3	-60.79	119.28	5.56	0.04
Sex + Age	3	-61.11	119.86	6.15	0.03
Global Model	8	-55.61	120.13	6.41	0.02
Sex + Age + Sex*Age	4	-61.09	122.00	8.28	0.01

[a] Number of parameters included in models. All models include an intercept, [b] Akaike Information Criterion adjusted for overdispersion and small sample size, [c] CCPN = proportion of closed-canopy pine within the winter home range, [d] ERSC = proportion of early successional habitat within the winter home range.

and techniques in our dispersal analysis. Whenever dispersal was detected, we calculated the Ac for all pre-dispersal (winter) locations and the Ac for all post-dispersal (breeding season) locations and buffered the centers by their respective 95% confidence limits. If the circles overlapped or if any post-dispersal location was within the pre-dispersal confidence limit circle, dispersal was rejected because we wished to adhere to the unidirectional definition of dispersal. The possibility of dispersal being detected on that particular set of n locations was then eliminated by setting a minimum distance to winter trap site for dispersal to be detected that was greater than the distance of the first n location to the trap site and the routine was repeated. Bobwhites trapped after 30 March (three days before the earliest recorded dispersal date, 2 April) were excluded from analysis because they could have already dispersed. Of the bobwhites not classified as dispersers, those that died or were censored prior to 19 May (the latest recorded date of dispersal) were excluded from analysis because they may not have had an opportunity to disperse. Of the bobwhites classified as dispersers, those that died before 16 days (the greatest number of days recorded between the initial date of an erroneous dispersal detection and a subsequent return) had elapsed since the initial dispersal date were also excluded from analysis because these birds may not have had time to return. Bobwhites are known to make long distance movements following nest failure (Urban 1972, Fies et al. 2002). We did not consider these to be dispersal events because those birds were already on their breeding range. To minimize the possibility of detecting these movements as dispersal events, dispersal detected after 19 May (the earliest recorded date of nest incubation) was not considered to be dispersal.

We used the animal movements extension (Hooge and Eichenlaub 1997) designed for ArcView® to calculate fixed kernel winter home ranges (Worton 1989) with a 95% isopleth requiring $\geq$20 locations to calculate home ranges. For bobwhites that dispersed, we only used pre-dispersal locations. For non-dispersers, we used locations collected before 15 April. We estimated ”pseudo home ranges” for birds with less than 20 winter locations by calculating the Ac for the locations that we did have and then buffering this point to create a 14.52

ha (average winter home range size) circle. We intersected home ranges with the habitat map of the study area using the ArcView® Geoprocessing Wizard to determine the proportions of home ranges composed of different habitat types.

We examined the effects of several covariates on spring dispersal probability using an information-theoretic approach. We developed an *a priori* set of 9 candidate models, based on literature review, our knowledge of bobwhite biology, and field observations. The model set included the effects of sex (coded Female = 0, Male = 1), age (Juvenile = 0, Adult = 1), proportions of closed-canopy pine plantations (CCPN) and early successional habitat (ERSC) within the winter home range, a sex*age interaction term, and terms for interactions between age and habitat covariates. We considered ERSC to be the most beneficial habitat and CCPN to be the most deleterious habitat for bobwhites in the landscape in which we were working (Lewis 1999, Parnell 2002, Cook 2004). Other habitat types were not considered because we viewed CCPN and ERSC as most likely to affect dispersal probability and we wished to minimize the number of models, thereby reducing the possibility of spurious results (Anderson and Burnham 2002). We included interactions between age and habitat because adults may perceive habitat quality differently due to their greater experience. We also included a null (intercept only) model in the candidate set. We used logistic regression (SAS PROC LOGISTIC; SAS Institute, Inc. 2002) to produce likelihood and parameter estimates for each model. We then calculated QAIC$_c$ (AIC corrected for small sample sizes and overdispersion) values and QAIC$_c$ weights for each model (Burnham and Anderson 2002). We assessed model fit using the Hosmer and Lemeshow (1989) goodness-of-fit statistic ($\hat{C}$).

We plotted model predicted dispersal probabilities with 95% confidence intervals over the range of habitat levels observed in the data for each age to evaluate the relationship of predicted dispersal probability over the range of our data. Model predicted dispersal probabilities were calculated using:

$$\text{Dispersal probability} = e^{ln(\hat{\pi})}/(1 + e^{ln(\hat{\pi})}),$$

where $\hat{\pi}$ is the model estimated logit.

Results

We radio-tagged 202 bobwhites from 39 coveys. We used 101 bobwhites in our dispersal analysis. Of these, 29.7% (4.6 SE) were classified as dispersers and mean dispersal distance was 1,835 m (194 SE) (Table 1). We excluded 8 birds from analysis because they were trapped after 30 March. We excluded 2 because they were classified as dispersers, but died before 16 days had elapsed since the initial dispersal date. We excluded 91 because they died or were censored prior to 19 May. Of these, 4 were censored during the dispersal period because radio contact was lost and not regained. Although transmitter failure could have caused the loss of radio contact, we may have been unable to find these birds because they dispersed beyond our search area. Therefore, we may have underestimated the dispersal rate and mean dispersal distance. If we did underestimate, we think that the underestimation was slight because only 4 birds were censored during this period. The selected best model ($\omega = 0.58$) for predicting spring dispersal included age, CCPN, and an age*CCPN interaction term (Table 2). This model fit the data ($\hat{C} = 0.15$). Age interacted with within winter home range proportions of CCPN ($\beta = 0.18$, 0.06 SE; Table 3). The null model was the next best model ($\omega = 0.16$). All other models performed poorly, ranking lower than the null model. Over the range of proportions of CCPN within winter home ranges (0-28%), predicted adult dispersal probability increased as the proportion of CCPN within the winter home range increased (Figure 1). Conversely, predicted juvenile dispersal probability decreased as the proportion of CCPN within the winter home range increased (Figure 2). However, confidence intervals for predicted dispersal probabilities were quite large for both age classes.

Discussion

Our results suggest that habitat may affect dispersal rates and this effect may vary by age. The

Table 3: Logistic regression parameter estimates of the selected best model (Age + CCPN + Age*CCPN) for predicting spring dispersal of radio-marked bobwhites monitored in Laurens County, Georgia during 2002-2003.

Parameter	**Coefficient**	**SE**	**LCI**[a]	**UCI**[b]
Intercept	-0.51	0.3	-1.1	0.07
Age	-1.14	0.66	-2.43	0.14
CCPN	-0.08	0.05	-0.18	0.01
Age*CCPN	0.18	0.06	0.05	0.3

[a] Lower 95% C.I. limit, [b] Upper 95% C.I. limit.

selected best model indicated that dispersal probability varied according to an interaction between age and proportions of closed-canopy pine within winter home ranges. Adults with greater proportions of closed-canopy pine within their winter home range were more likely to disperse. Because of greater experience, adults may perceive habitat differently from juveniles, which could influence adult tendency to disperse. Surprisingly, model predictions suggested that juvenile dispersal probability may decrease with increasing proportions of within winter home range closed canopy pine. Juveniles could perceive closed-canopy pine as quality woody cover. There is evidence that closed-canopy pine plantations are especially deleterious to bobwhite survival (Parnell 2002, Cook 2004); therefore, they may serve as an ecological trap to juveniles. However, we suggest that predictions based on this model be interpreted cautiously. We did not test this model with independent data. Although the age*CCPN interaction was statistically significant, confidence intervals for predicted dispersal probabilities were quite broad; therefore, the magnitude of the effect remains uncertain. Because of this and the apparently negligible effect of early successional habitat on dispersal probability, it is likely that a significant portion of birds from both age classes could disperse every spring, regardless of habitat quality. However, only 16% of birds in this study had winter home ranges comprised of >50% early successional habitat (range 2-82%). At higher proportions, early successional habitat may have a greater effect on dispersal probability. Although this is the first study to report an age-habitat interaction effect on bobwhite dispersal probability, other researchers have suggested that habitat affects bobwhite dispersal patterns. Urban (1972) reported that sizeable weed areas within winter home ranges tended to prevent bobwhites from shifting their home ranges. Duck (1943) attributed a shift from fall to winter ranges by bobwhites to a change in habitat preference. Of course, differences in matrix habitat may affect dispersal distances as well (Turner et al. 2001, pg. 220). Puckett et al. (1995) reported that distances from capture site to first nest for bobwhites were over four times greater on areas without herbaceous filter strips (beneficial habitats) versus areas with them. Fies et al. (2002) proposed an inverse relationship between dispersal distances of bobwhites and inter-patch connectivity which is supported by reports that mobility is lower on areas managed intensively for bobwhites (Loveless 1958, Smith et al. 1982, Terhune et al. 2006) and greater on areas containing marginal habitat (Kabat and Thompson 1963, Fies et al. 2002, this study). The effect of habitat on bobwhite dispersal probability may not be responsible for lower mobility on managed areas. We detected little effect of early successional habitat on dispersal probability. Perhaps bobwhites disperse in all landscapes, but dispersal distances vary according to inter-patch connectivity.

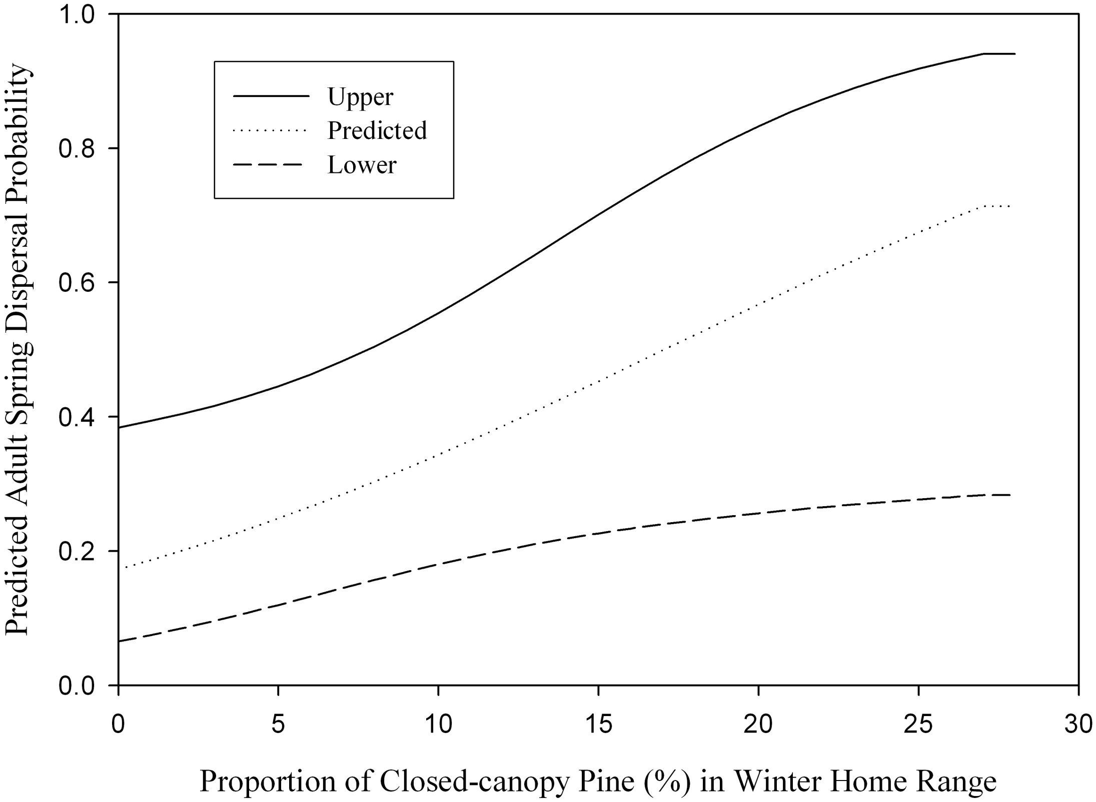

Figure 1: Predicted spring dispersal probability for adult bobwhites with 95% confidence intervals based on the selected best logistic regression model (Age + CCPN + Age*CCPN) for predicting spring dispersal of radio-marked bobwhites monitored in Laurens County, Georgia during 2002-2003.

Dispersers may be exposed to greater predation risk due to increased movement (Ambrose 1972, Smith 1974) and because dispersers inhabit space where they are unfamiliar with cover and food resources (Clark et al. 1993, Jacquot and Solomon 1997, Yoder et al. 2005). Researchers have reported that survival rates of dispersed bobwhites are either identical to or even greater than that of non-dispersed bobwhites (Townsend et al. 2003, Cook 2004, Mike Fies, Virginia Dept. of Game and Inland Fisheries, unpublished data). However, mortality that occurs during dispersal (transience) has yet to be examined. Dispersers may experience increased mortality during transience, especially in a hostile landscape matrix (Fahrig 2001). Examining the effect of daily movement rate on bobwhite survival, Folk (2006) reported that individuals that moved >890m in a day were twice as likely to be killed as birds that did not move at all. Although not directly measured in that study, the finding does suggest that bobwhite transience survival may be quite low in some landscapes. More research is needed to determine bobwhite mortality during transience and how this may be affected by matrix habitat composition and configuration.

Management Implications

Although dispersal may allow bobwhite populations to persist in fragmented landscapes, efforts to increase populations at the local scale (e.g. public wildlife management areas) are hindered if emigration greatly exceeds immigration. Our results suggest that habitat features may influence dispersal probability. However, the magnitude of the ef-

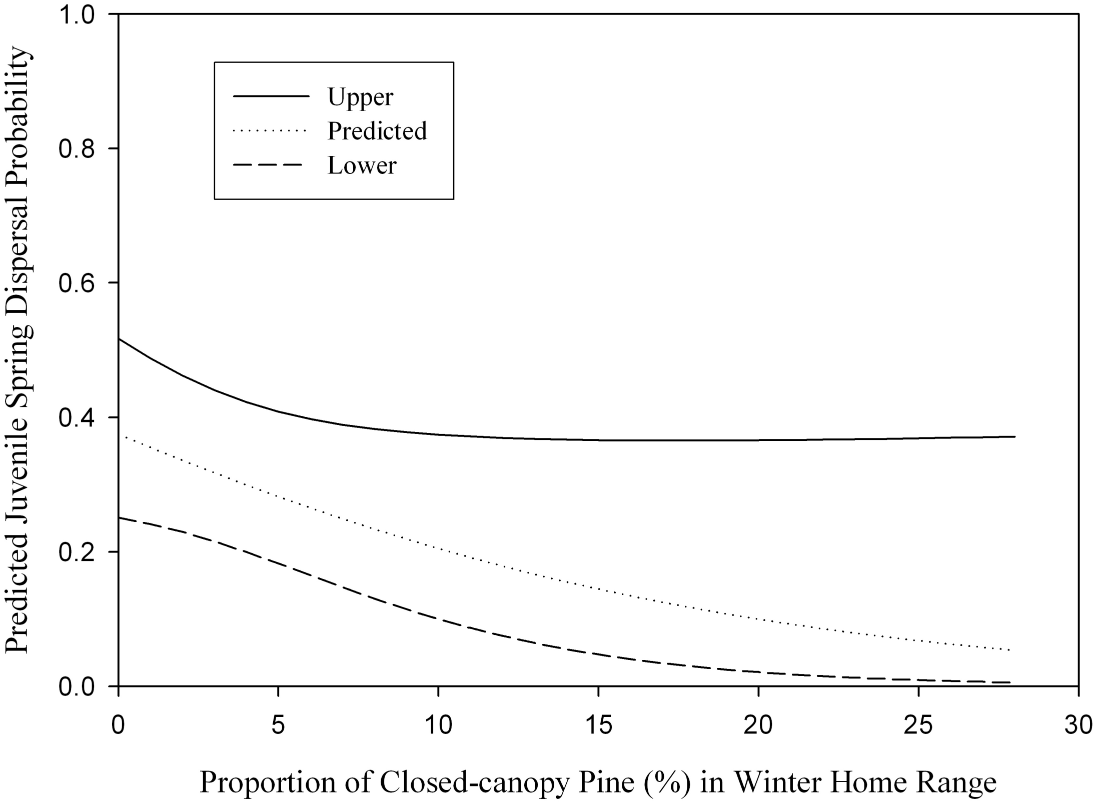

Figure 2: Predicted spring dispersal probability for juvenile bobwhites with 95% confidence intervals based on the selected best logistic regression model (Age + CCPN + Age*CCPN) for predicting spring dispersal of radio-marked bobwhites monitored in Laurens County, Georgia during 2002-2003.

fect of closed-canopy pine plantations on dispersal probability is uncertain and the amount of early successional habitat within winter home ranges appeared to have little effect on dispersal probability. It is likely that a significant portion of birds will disperse every spring, regardless of habitat quality on a management unit. Therefore, it is important to consider surrounding landscape quality and management unit size when determining which areas are most likely to respond to management and the proper management strategy needed to achieve bobwhite population objectives. Lower surrounding landscape quality (amount of and proximity to suitable habitat) will likely result in lower immigration rates. Smaller management unit size will likely result in lower immigration and higher emigration because birds on the management unit and surrounding areas that disperse will be less likely to form breeding ranges on the management unit. Because bobwhites are known to select for early successional habitat, immigration should increase as the amount of early successional habitat increases on an area. However, surrounding landscape quality and management unit size will still affect immigration rates. As surrounding landscape quality and management unit size decrease, managers must increase the intensity of their management to achieve bobwhite population objectives. On many areas in the modern landscape, managers may have to adopt an "all out" management strategy to offset losses to emigration and achieve bobwhite populations large enough to support hunting. This type of strategy would include converting all available acreage to bobwhite habitat and, in some cases, adopting practices that may be considered controversial (e.g. predator control). Not adopting this type of management strategy may lead to unrealized objectives in many cases. Finally, the greater mobility reported in this and similar studies conducted in fragmented landscapes (Kabat and Thompson 1963, Fies et al. 2002) compared to the lower mobility reported by studies conducted on areas of contiguous habitat (Loveless 1958, Smith et al. 1982, Terhune et al. 2006) strongly suggests that bobwhites disperse greater distances in fragmented landscapes. How exactly metapopulation theory fits bobwhites has yet to be determined, but many metapopulation principles are almost certainly applicable given the relatively low mobility of the species and the fact that it often inhabits fragmented landscapes (Burger 2002, Fies et al. 2002). In these fragmented landscapes, both dispersal rates and distances will determine the rates of emigration and immigration between populations and ultimately the long term viability of regional populations (Hanski 1999). Fies et al. (2002) recommended that areas of suitable habitat should be located within a ″yet-to-be-defined critical dispersal distance.″ More information on how landscape attributes affect bobwhite dispersal patterns and transience survival is needed before this distance can be defined.

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Usable Space Versus Food Quantity in Bobwhite Habitat Management

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We studied the response of northern bobwhite (*Colinus virginianus*) foods (plants and invertebrates), usable space, and populations following thinning and burning on the 60,000-ha pine (*Pinus* spp.)-grassland restoration area in the Ouachita National Forest, Arkansas, to examine 2 hypotheses commonly used to manage bobwhite habitat: 1) usable space (suitable permanent cover) and 2) food quantity (an element of habitat quality). We estimated invertebrate food abundance using sweep nets and abundance of food-producing plants using herbaceous and woody stem counts. The disk of vulnerability was used to index usable space. We used whistling-male counts to index population response. Relative abundance, mass, and frequency of occurrence of invertebrate foods and richness, density, and frequency of occurrence of bobwhite food-producing plants increased following thinning and fire. Relative abundance of whistling males was greatest in thinned stands 3 growing seasons post-burn and in thinned but unburned stands. We found food supply was related to usable space following treatment. However, food abundance alone did not explain bobwhite population response, whereas, usable space was predictive for bobwhite response. By comparing treated stands with similar usable space but different food quantity, we observed no differences in bobwhite abundance. Neural models suggested bobwhite population response was less sensitive to changes in food supply relative to changes in usable space. We recommend that managers should seek first to provide usable space (suitable permanent cover in low basal area stands), recognizing that adequate food supply will likely be a side effect of management to this end.

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Key words: Arkansas, *Colinus virginianus*, food quantity, management philosophies, northern bobwhite, Ouachita Highlands, pine-grassland restoration, prescribed fire, usable space

Introduction

Currently, northern bobwhite habitat managers have 2 hypotheses from which to choose when considering management programs. The usable space hypothesis formalized by Guthery (1997) contends as suitable habitat increases on an area of fixed size, mean bobwhite density will increase on the area. Usable space can be defined as suitable permanent cover. The second hypothesis predicts bobwhite density is a function of food quantity (Guthery 1997). This hypothesis contends habitat quality, such as food supply, exists along a continuum ranging from poor to good. Bobwhite management practices such as food plots and food supplementation operate under the quality hypothesis. Any number of habitat variables such as water supply, thermal cover, or habitat-type interspersion could be considered measures of habitat quality. Quality-based management assumes a higher level of habitat quality will support a greater number of bobwhites.

The food quantity hypothesis assumes food is limiting in a given area and increasing the food supply with food plots or supplemental feeding will increase bobwhite densities. We contend managers often focus first on addressing the quantity of the food supply rather than usable space. If food is assumed limiting in a given area, literature reporting on food-increasing management techniques should

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indicate an effect on bobwhite densities commensurate with an increase in food supply (Guthery 1997, 2002). However, the literature suggests management techniques aimed at increasing food supplies are ineffective in terms of increasing fall bobwhite densities (Burger and Linduska 1967, Ellis et al. 1969, Guthery 1997, 2002, Guthery et al. 2004).

Bobwhite populations have responded positively across the southeastern United States in forest stands managed for the endangered red-cockaded woodpecker (*Picoides borealis*) (Brennan 1991, Fuller 1994, Wilson et al. 1995). In Arkansas, bobwhites were found more frequently in thinned and burned stands managed for the red-cockaded woodpecker than in unthinned and unburned control stands (Wilson et al. 1995, Cram et al. 2002). To manage for the red-cockaded woodpecker on the Ouachita National Forest the U.S. Forest Service has delineated a 60,000 ha area for pine-grassland ecosystem restoration. Pine-grassland restoration efforts in the Ouachita National Forest included a program of tree thinning called wildlife stand improvement (WSI) and dormant-season prescribed fire every 3 years. WSI removes <1/3 of the overstory shortleaf pine (*P. echinata*) and >2/3 of the hardwood midstory, and has created stand structure with an open midstory maintained by fire (Wilson et al. 1995, Masters et al. 1998).

Our objective was to investigate the usable-space hypothesis (Guthery 1997, 2002) versus the food quantity-based hypothesis to determine if either or both explained an increase in bobwhite relative abundance. We predicted that food supply, both plant and animal, would increase through the 3 growing seasons following midstory removal and fire, yet would have nominal effects in terms of an increase in relative bobwhite abundance as compared to an increase in usable space (suitable permanent cover). An increase in usable space, as determined by an increase in relative bobwhite abundance, was defined by an increase in forest hectares restored to open pine-grassland conditions following restoration treatment (i.e., WSI and dormant season prescribed fire every 3 years) (Cram et al. 2002).

Study Area

Study sites were in the west-central Ouachita Mountains on the Poteau Ranger District of the Ouachita National Forest, Scott County, Arkansas. All sites were within the 60,000-ha pine-grassland restoration area and under active management for the endangered red-cockaded woodpecker. The Ouachita Mountains cover an area approximately 380 km east to west by 100 km north to south in western Arkansas and southeastern Oklahoma. Mountain ridges typically run east-west with long north-facing and south-facing slopes. The drier south-facing slopes characterized study sites. Elevations range from 100 to 900 m.

The forest is composed of mixed pine-hardwood stands with shortleaf pine dominating drier south-facing slopes, and hardwoods (primarily oaks [*Quercus* spp.] and hickories [*Carya* spp.]) dominating mesic north-facing slopes (Foti and Glenn 1991). Codominant overstory and midstory species included red maple (*Acer rubrum*), mockernut hickory (*C. tomentosa*), pignut hickory (*C. glabra*), flowering dogwood (*Cornus florida*), black cherry (*Prunus serotina*), Mexican plum (*P. mexicana*), southern red oak (*Q. falcata*), blackjack oak (*Q. marilandica*), northern red oak (*Q. rubra*), post oak (*Q. stellata*), and black oak (*Q. velutina*). Post oak, blackjack oak, red maple, and mockernut hickory sprouts <3 m tall dominated the understory in WSI stands 3 years postburn. Woody shrub and vine species included New Jersey tea (*Ceanothus americanus*), blackberry (*Rubus* spp.), Virginia creeper (*Parthenocissus quinquefolia*), winged sumac (*Rhus copallina*), greenbrier (*Smilax bona-nox*), poison ivy (*Toxicodendron radicans*), low-bush huckleberry (*Vaccinium pallidum*), and muscadine (*Vitis rotundifolia*) (Sparks 1996).

Methods

Experimental Design

We used a completely randomized design over 2 years with 4 replications of 5 treatments in 20 stands in 1999 and 2000 for a total of n = 40 stands. Each year 20 stands ≥16 ha ($\bar{x} \pm$ SE; 35 ha ± 2.9) were randomly selected from a list of all suitable stands in the restoration area. Treatment stands in 1999 and 2000 were stratified based on the number of 3-year burning cycles completed (1-7). Treatments (n = 8 for each treatment) were 1) unthinned, unburned control; 2) WSI-no burn (WSI-NB); 3) WSI-burn, first growing season after dormant-season burn (WSI-B1); 4) WSI-burn, second growing season after dormant-season burn (WSI-B2); 5) WSI-burn, third growing season after dormant-season burn (WSI-B3).

Bobwhite Counts

To estimate bobwhite abundance we used whistling-male call counts with playback recordings (Coody 1991) at 1-2 listening points/stand over a 2-week period in May 1999 and 2000. Points were centrally located $\geq$200 m from stand edge. Each point had an implied 200-m radius of audibility contrary to the standard 400-m radius (Stoddard 1931, p. 102) of rangelands because topography effects on the ONF reduced the distance sound waves could be detected by a human. Whistle counts were repeated 3 times by 3 different individuals between sunrise and 1100 hrs. Whistle counts were stratified during the morning to encompass peak calling periods.

We recorded the number of different whistling males over a 6-min listening period. Playback of an assembly call (Don Scott, Lake Charles, Louisiana, USA) broadcast at 90 dB in the cardinal directions was used twice, once at the 3-min mark and again after the 4.5-min mark (Coody 1991). Relative abundance as indexed by whistle counts is reported by treatment as mean whistling males/point.

Covey-call counts were conducted 3 times by 3 different observers 45 mins before sunrise to 1100 hrs during the first week in October 1999 and 2000. Listening-point locations and assembly-call broadcast methodology were unchanged from whistle-count procedures. The 6-min listening periods were stratified by observer to encompass peak calling times. We recorded the number of different calling coveys and reported relative abundance by treatment as mean coveys/point.

Invertebrate Sampling

To index invertebrate abundance during critical brood-rearing months (June-August) (Stoddard 1931, Rosene 1969, pp. 41 and 59, respectively), we examined the effects WSI and fire had on invertebrate abundance, mass, and frequency of occurrence in untreated pine-hardwood stands as compared to treated stands at various stages of succession following thinning and burning. We collected invertebrates using a standard canvas sweepnet (48-cm handle, 38-cm net hoop diameter, and 76-cm net depth) to estimate relative abundance, mass, and percent frequency of occurrence. Invertebrate sweepnet samples were collected in each stand along 6 randomly located transects 25 m in length on 2 randomly spaced parallel lines (i.e., 3 transects per line), perpendicular to the contour. We used 20 sweepnet strokes/transect line. Transect lines bisected bobwhite whistle-call sampling points. Invertebrates were collected in July 1999 and 2000 between 1000 and 1500 hours when cloud cover was <50% and temperatures were <35° C. Contents of sweepnets were transferred to labeled plastic bags, sealed, and frozen for storage. Invertebrates were sorted to order following Borror et al. (1989), dried at 40° C for 72 hours, and weighed to the nearest 0.001 g. Relative invertebrate abundance and mass were calculated from the 6 transect samples, and reported as mean individuals/sample and mean mg/sample. Percent frequency of occurrence was calculated for the 6 transects.

Sweepnet sampling was selected because of its widespread acceptance as an invertebrate sampling technique (Callahan et al. 1966). Although shortcomings associated with sweepnet sampling are acknowledged (Thompson 1987), sweepnet samples do reflect the taxonomic heterogeneity and magnitude of the invertebrate biomass present in the vegetative canopy of grasslands (Evans et al. 1983). Time and resource constraints precluded the use of vacuum sampling. Vacuum sampling is potentially better suited to trap invertebrates more vulnerable to chick foraging, i.e., invertebrates that are small in size, on the ground, and relatively slow moving (compared to aerial invertebrates).

Vegetation Sampling

To examine plant food quantity, we estimated density and frequency of occurrence of known bobwhite food-producing plants based on regional food habit studies (Baumgartner et al. 1952, Masters et al. 1996, Bidwell et al. 1998). To characterize and index bobwhite food-producing plants in each stand, we sampled 30 1-m^2 plots at 30-m intervals on 2-4 randomly spaced parallel lines, perpendicular to the contour over a 2-week period in July 1999 and late-June 2000. We recorded density for each herbaceous species within plots. We recorded density for woody vegetation within 30 fixed-radius plots (radius 3.59 m). We divided woody understory, shrub, and midstory species into 3 height classes: 0-1, $>$1-3, and $>$3 m. To further index structure we estimated the disc of vulnerability (Kopp et al. 1998) by measuring the distance at which a 15 X 2.5 cm cylinder disappeared from view of a kneeling observer (height = 1 m) at cardinal radii, then used mean distance to calculate area. A comprehensive list of individual bobwhite food-producing species counted on the Ouachita National Forest was reported in Cram (2001). To avoid bias from surrounding stands, no sampling was conducted within 50 m of stand edge (Mueller-Dombois and Ellenberg 1974, p. 123).

Data Analysis

We calculated species richness of bobwhite food-producing herbaceous and woody vegetation at the stand level. We summarized herbaceous and woody species by mean density and percent frequency of occurrence for each treatment. Differences in means between years and treatments were tested using Kruskal-Wallis nonparametric tests (Steel et al. 1997, p. 177). Stand (year x treatment) Type III mean square was the error term (SAS Institute, Inc. 1985, p. 651). We used multiple comparisons between mean ranks with the Least Significant Difference (LSD) test with P = 0.050 (Steel et al. 1997, p. 178). Stand means were tested for homogeneity of variance among treatments using Levene's test (Snedecor and Cochran 1980). Regression analysis was used to examine relationships among total plant food abundance and invertebrate abundance and mass with whistle-count results.

To further understand nonlinear effects, we modeled mean whistling-male response to habitat variables using artificial neural-network models. Neural Connection software (SPSS Inc., Chicago, Illinois, USA) was used to conduct modeling. We used neural models to detect relationships between mean whistling-male abundance and habitat structure and composition following treatment. Our model used 6 input nodes (independent variables), 1 hidden node, and 1 output node (dependent variable). The *a priori* rationale for choosing 1 hidden node was to prevent overtraining, which would result in models that generalize poorly. The input nodes were year and stand means for forb cover, preferred bobwhite invertebrate abundance, hardwood basal area, conifer basal area, and exposure to ground predators (disc of vulnerability) (Cram et al. 2002). The output node was predicted whistling males/point. The neural model was trained using a randomly drawn data set comprising 80% of the data (n = 32); testing was conducted on the remaining 20% of the data (n = 8).

Results

Population Response

Based on spring whistle counts, the greatest relative abundance of bobwhites occurred in unburned, thinned stands ($\bar{x} = 1.1 \pm 0.32$ [SE]) and in thinned stands in the third growing season following fire ($\bar{x} = 1.54 \pm 0.39$ [SE]) (Cram et al. 2002). Thinned stands in the first (WSI-B1) and second (WSI-B2) growing seasons following fire had similar levels of relative bobwhite abundances ($\bar{x} = 0.4 \pm 0.2$ [SE], $\bar{x} = 0.8 \pm 0.3$ [SE], respectively) (Cram et al. 2002). Control stands had the least measure of bobwhite relative abundance ($\bar{x} = 0.1 \pm 0.1$ [SE]) (Cram et al. 2002). There was no statistically significant difference in mean bobwhite relative abundance between 1999 and 2000 (1999: $\bar{x} = 1.0 \pm 0.2$ [SE], 2000: $\bar{x} = 0.6 \pm 0.2$ [SE], P = 0.157). Based on covey-call counts, relative abundance of covey calls was similar in nature to whistle counts; relative abundance of covey calls was greatest in unburned, thinned stands (WSI-NB) ($\bar{x} = 0.50$ ± 0.27 [SE]) and in thinned stands 3 growing seasons following fire (WSI-B3) ($\bar{x} = 0.57$ ± 0.30 [SE]) (Cram et al. 2002). No coveys were detected in control stands using covey-call counts (Cram et al. 2002).

Invertebrate Response

Relative invertebrate abundance (mean invertebrates/sample) and mass (mean mg/sample) increased over control stands following WSI and fire treatment (Table 1). Thinned stands in the third growing season following fire had the greatest total invertebrate abundance and mass as compared to other treatments. Total invertebrate abundance was more than 2-fold greater than controls and total invertebrate mass was more than 3-fold greater than controls in WSI stands 3 growing seasons following fire. Relative to the total number of invertebrate orders identified (12) there were few differences between orders between years in terms of relative abundance (i.e., Araneae, Homoptera, and Lepidoptera differed between years) or mass (i.e., Homoptera, and Lepidoptera differed between years) of individual orders, but no differences between years when total abundance or total mass was considered.

Sweepnet sampling captured invertebrates from 12 different orders (see Cram 2001) with locomotion adaptations ranging from cursorial to saltatorial to aerial. Invertebrates frequently consumed by bobwhite adults and chicks included Coleoptera, Hemiptera, Homoptera, Lepidoptera larvae, and Orthoptera (Stoddard 1931, Hurst 1972, Jackson et al. 1987). Percent frequency of occurrence of these important invertebrate orders increased following thinning and fire (Table 2). Orthoptera had 100% frequency of occurrence in WSI-B3 stands. Araneae, Coleoptera, Homoptera, Lepidoptera larvae, and Orthoptera abundance were all positively related to number of times a stand had been burned (*r* = 0.32, 0.44, 0.49, 0.63, 0.52, respectively).

Herbaceous and Woody Response

Of 286 different herbaceous and woody species identified using stem counts on the Ouachita National Forest, 52 (18%) herbaceous and 14 (5%) woody species were known to be food-producing plants for bobwhites and used in data analysis. Orthogonal contrasts indicated 22 herbaceous and 5 woody species increased in density following thinning and burning as compared to controls. Herbaceous species richness of bobwhite foods was greatest in thinned and burned stands 1, 2 and 3 growing seasons following fire (Table 3). Total herbaceous stems (stems/m^2) were greatest following fire and decreased 2 and 3 growing seasons following fire (Table 3).

Total panicum species (*Panicum* spp.), a preferred bobwhite food in pine-oak forests (Baumgartner et al. 1952), increased following thinning and maintained higher densities than controls following fire (Table 2). Percent frequency of occurrence of wooly panicum (*P. acuminatum*), Bosc panicum (*P. boscii*), forked panicum (*P. dichotomun*), open-flower panicum (*P. laxiflorum*), and slimleaf panicum (*P. linearifolium*) all increased following thinning and again following burning.

We identified 25 different species of legumes, including 10 species of tick trefoil and 7 species of bush clover. Total legume stems (stems/m^2) increased >3-fold 1, 2, and 3 growing seasons following fire (Table 2). Hog peanut (*Amphicarpaea bracteata*), partridge pea (*Cassia fasciculata*), and downy-milk pea (*Galactia regularis*), preferred legumes by bobwhites (Baumgartner et al. 1952), increased in density in WSI treated stands as compared to control stands. We found 13 legume species increased in percent frequency of occurrence in response to fire alone. Densities of partridge pea (*Cassia fasciculata*), butterfly pea (*Clitoria mariana*), small-leaved trick trefoil (*D. ciliare*), beggar's lice (*D. laevigatum*), panicled trick trefoil (*D. paniculatum*), tick trefoil spp., tick trefoil (*D. viridiflorum*), downy-milk pea (*Galactia regularis*), bicolor lespedeza (*Lespedeza bicolor*), prostrate lespedeza (*L. procumbens*), and reclining lespedeza (*L. repens*) were positively related to number of times

Table 1: Relative invertebrate mass (mean mg/sample) and invertebrate relative abundance (mean invertebrates/sample) response to wildlife stand improvement and fire on the Ouachita National Forest, Arkansas, July 1999 and 2000.[a]

	Treatment[b]										
Variable	Control	SE	WSI-NB	SE	WSI-B1	SE	WSI-B2	SE	WSI-B3	SE	$P > F$
Total mass	53.3	19.2 C	79.8	15.6 BC	84.1	21.4 BC	132.4	21.6 AB	166	28.9 A	0.004
Total abundance	11.1	1.9 C	14.6	1.6 C	17.5	3.1 BC	21.2	2.4 AB	26.2	3.6 A	0.004

[a] Row means followed by the same letter or without letters were not significantly different at the 0.05 level (LSD)

[b] Control = unthinned, unburned; WSI-NB = wildlife stand improvement, no burn; WSI-B1 = wildlife stand improvement, second growing season following burn; WSI-B3 = wildlife stand improvement, thrid growing season; n = 8 for each treatment

Table 2: Invertebrate (only orders frequently consumed by northern bobwhites) percent frequency of occurrence response to wildlife stand improvement and fire on the Ouachita National Forest, Arkansas, July 1999 and 2000.[a]

	Treatment[b]										
Invertebrate order	Control	SE	WSI-NB	SE	WSI-B1	SE	WSI-B2	SE	WSI-B3	SE	$P > F$
Coleoptera	43.8	9.9 B	54.2	7.6 B	77.1	8.9 A	81.3	5.8 A	85.7	7.7 A	0.003
Hemiptera	2.1	2.1 B	56.3	8.3 A	62.5	8.8 A	62.5	5.2 A	76.2	8.8 A	< 0.001
Homoptera	18.8	6.6 B	56.3	10.9 A	60.4	10.4 A	66.7	13.7 A	76.2	13.5 A	0.012
Lepidoptera larvae	16.7	5.5 D	35.4	5.8 CD	39.6	8.9 BC	60.4	9.4 AB	64.3	5.7 A	< 0.001
Orthoptera	56.3	8.3 C	83.3	6.3 B	87.5	6.9 AB	97.9	2.1 A	100	0.0 A	< 0.001

[a] Row means followed by the same letter or without letters were not significantly different at the 0.05 level (LSD)

[b] Control = unthinned, unburned; WSI-NB = wildlife stand improvement, no burn; WSI-B1 = wildlife stand improvement, first growing season following burn; WSI-B2 = wildlife stand improvement, second growing season following burn; WSI-B3 = wildlife stand improvement, third growing season following burn; n = 8 for each treatment

Table 3: Herbaceous (stems/m^2) and woody (thousand stems/ha) northern bobwhite food-producing stem category response to midstory thinning and dormant-season prescribed fire on the Ouachita National Forest, Arkansas, July 1999 and 2000.[a]

	Treatment[b]										
Category[c]	Control	SE	WSI-NB	SE	WSI-B1	SE	WSI-B2	SE	WSI-B3	SE	$P > F$
Total stems											
Grass[d]	0.3	0.2	0.2	0.2	0.6	0.5	0	0	0	0	0.453
Panicum	2.5	0.6 C	18.5	4.2 B	24.9	3.2 AB	26.6	3.1 A	24.9	5.7 AB	< 0.001
Sedge	2.2	0.3 B	8.2	1.4 A	7.8	2.9 A	9	2.1 A	8.2	3.0 A	0.001
Legume	4.3	1.0 B	5.3	0.9 B	25.3	6.1 A	22.3	4.3 A	19	2.2 A	< 0.001
Forb	0.4	0.2 C	2.2	0.5 B	13.5	4.0 A	7.6	1.3 A	7.4	1.4 A	< 0.001
All herbaceous	9.7	1.7 C	34.5	5.2 B	72.1	8.8 A	65.5	6.1 A	59.5	8.5 A	< 0.001
All woody	23.1	3.8	20.7	3.5	36.6	7.7	17	3.7	16.7	3.4	0.125
Species richness											
Herbaceous	16.5	1.5 C	23.1	1.0 B	29.9	1.6 A	28.4	1.4 A	27.6	0.9 A	< 0.001
Woody	10.3	0.3	11.9	0.6	11.6	0.8	10.6	0.8	11.9	0.6	0.079

[a] Row means followed by the same letter or without letters were not significantly different at the 0.05 level (LSD)

[b] Control = unthinned, unburned; WSI-NB = wildlife stand improvement, no burn; WSI-B1 = wildlife stand improvement, first growing season following burn; WSI-B2 = wildlife stand improvement, second growing season following burn; WSI-B3 = wildlife stand improvement, third growing season following burn; n = 8 for each treatment

[c] Food-producing stems only

[d] Not including panicum stems

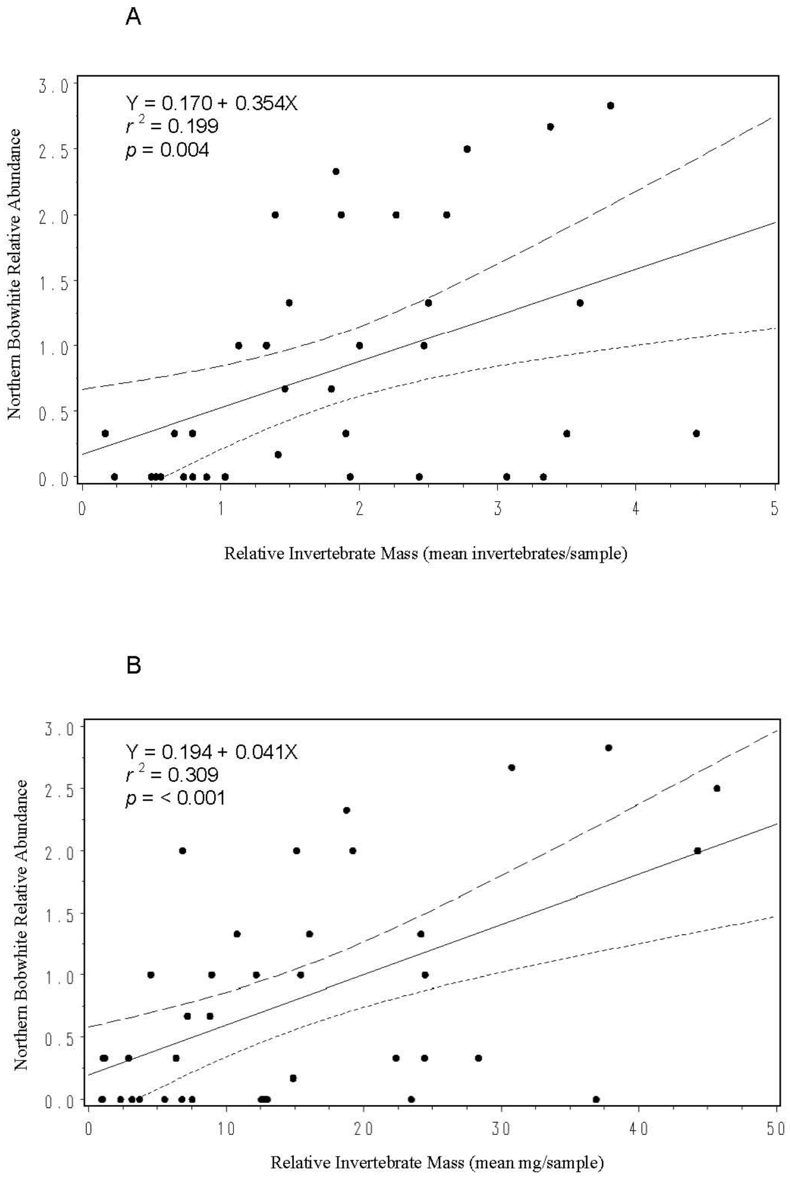


Figure 1: Response of northern bobwhite whistling males (mean whistling males/point) to increasing relative invertebrate abundance (mean invertebrates/sample) (n = 40) of Coleoptera, Hemiptera, Homoptera, Lepidoptera larvae, and Orthoptera (A), and to increasing relative invertebrate mass (mean mg/sample) (n = 40) of Coleoptera, Hemiptera, Homoptera, Lepidoptera larvae, and Orthoptera (B) on the Ouachita National Forest, Arkansas, July 1999 and 2000 (95% confidence intervals shown with dashed lines).

burned ($0.33 < r < 0.70$).

Total forb stem density (stems/m^2) increased also after thinning and again following fire (Table 3). Preferred forbs, common ragweed (*Ambrosia artemisiifolia*), and rough-leaf sunflower (*Helianthus hirsutus*), increased in density following WSI treatment. Three-seeded mercury (*Acalypha gracilens*), plains tickseed (*Coreopsis tinctoria*), rough-leaf sunflower, and black-eyed susan (*Rudbeckia hirta*) increased in percent frequency of occurrence following fire treatment.

Total woody stems (stems/m^2) were greatest following fire and decreased 2 and 3 growing seasons following fire (Table 3). Winged sumac, smooth sumac (*R. glabra*), and farkleberry (*Vaccinium arboreum*) increased in density in response to thinning and again in response to fire. Winged sumac, smooth sumac, and blackberry increased in percent frequency of occurrence following WSI.

Bobwhite Response to Food Abundance

We found increases in abundance and mass of frequently consumed invertebrates explained 20% and 31% of the variation in bobwhite relative abundance (Figure 1). No strong relationships were detected between total stems of grass, panicum, legume, or forb with bobwhite relative abundance. Linear regression indicated an increase in total bobwhite food-producing herbaceous stems explained only 15% of the variation in bobwhite relative abundance (Figure 2). The neural model explained 40% of the variation in the training data and 32% of the variation in the validation data. Bobwhite relative abundance appeared more sensitive to a decrease in disc of vulnerability as compared to increases in forb cover or preferred bobwhite invertebrate abundance (Figure 3).

Discussion and Conclusions

Hypothesis Testing

The preeminent dichotomy in bobwhite habitat management remains managing for food quantity or usable space. Guthery et al. (2001) indirectly tested the habitat quantity versus quality hypothesis and found bobwhite abundance increased with usable space on areas of fixed size, and declined with Shannon diversity of patch types, patch richness, and woody edge density (as they defined it). However, >70% of the variation in bobwhite abundance remained unexplained by the usable space hypothesis. In a *post facto* comparison between the 2 philosophies Taylor et al. (1999) also found ambiguous results.

We found the effects of increased food supply (invertebrate abundance and mass, and herbaceous food stems) following thinning and fire on bobwhite relative abundance were ambiguous in terms of supporting either the usable space hypothesis or the food quantity hypothesis. Because bobwhite abundance increased as a function of usable space (Cram et al. 2002) and bobwhite abundance increased somewhat as a function of food supply (Figs. 1, 2), food supply and usable space were confounded; food supply may be a function of the usable space created following pine-grassland restoration. However, food is not a condition of the usable space hypothesis and therefore food abundance cannot create usable space per se (Guthery 1997).

Deductions, however, can be made to separate the correlated effects of usable space and food supply. WSI-NB and WSI-B3 stands had similar amounts of usable space as measured by the mean disc of vulnerability (m^2) ($\bar{x}$ ± SE; 75.8 ± 14.8, 52.0 ± 7.7, respectively) and woody stem density (stems/plot) ($\bar{x}$ ± SE; 126.2 ± 15.7, 161.5 ± 21.9, respectively) and measures of bobwhite relative abundance (bobwhite/ha) ($\bar{x}$ ± SE;1.1 ± 0.3 and 1.5 ± 0.4, respectively), but significantly different food supplies as measured in preferred invertebrate abundance, mass, and herbaceous stem density of food-producing plants (Tables 1, 2, 3). The food quantity hypothesis contends an increase in food supply should result in an increase in bobwhite abundance, while the usable space hypothesis contends a threshold in the food supply has been met and no further increase in food supply will result in an increase in bobwhite abundance. Based on this observation, we deduced bobwhites responded to an increase in usable space rather than an increase in food supply, or conversely, food was not limiting following thinning and burning.

Artificial neural network model predictions were consistent with this deduction. Changes in habitat structure, predominately woody cover <2 m as indexed by the disc of vulnerability, largely predicted whistling male abundance. A threshold region appeared to exist beyond which the addition of increased food resources had a minor effect on bob-

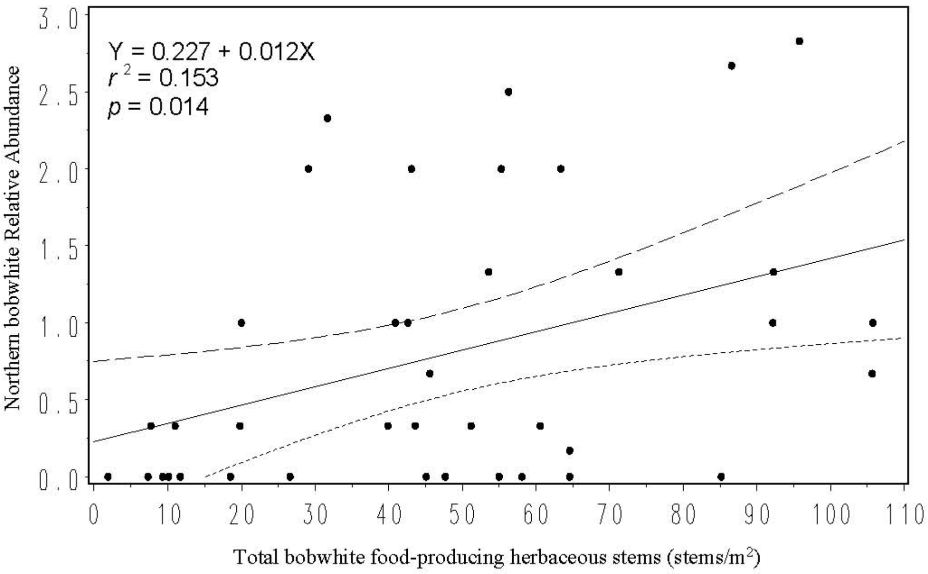
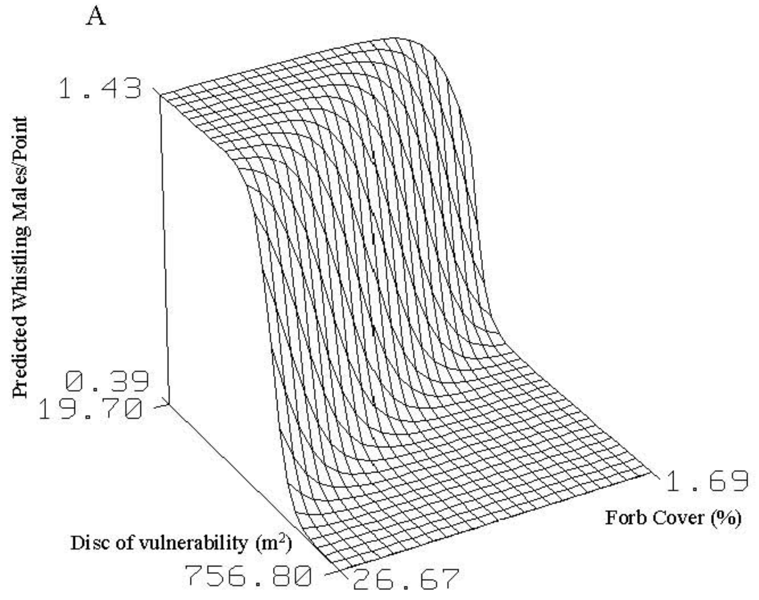


Figure 2: Response of northern bobwhite whistling males (mean whistling males/point) to increasing total bobwhite food-producing herbaceous stems (stems/m^2) ($n = 40$) on the Ouachita National Forest, Arkansas, July 1999 and 2000 (95% confidence intervals shown with dashed lines).

white abundance (Figure 3). Furthermore, Palmer et al. (2001) found greater densities of invertebrates in a defined area did not translate linearly into greater benefits to bobwhite chicks as indexed by foraging rate or a growth index.

Guthery (1999) offered a hypothesis explaining the general circumstance: food supplies as evaluated through energy-based carrying capacity routinely exceed the needs of bobwhite populations. Furthermore, the literature on the effects of food plots and food supplementation has failed to provide unchallengeable evidence an increase in food supply results in positive bobwhite population response as measured by fall densities (Guthery 1997, 2002). It has also been argued (Palmer et al. 2001) that a problem may exist in equating food supply directly to available food. However, an ongoing study of bobwhite food habits on the same study areas in Arkansas (R. E. Masters, unpublished data) suggests that the food supply items measured were consumed and ranked high in preference. Therefore as measured in this study, the increase in frequency of occurrence in herbaceous species and preferred bobwhite invertebrates following thinning and fire suggests an increase in bobwhite food availability. Frequency of occurrence provides an indication of uniformity in distribution (Mueller-Dombois and Ellenberg 1974). Although sweepnets may have missed exclusively cursorial invertebrates important to chick survival, arguably, we assumed these invertebrates responded in similar fashion to habitat change as compared to captured orders. Southwood (1968) and Southwood et al. (1979) reported the most prominent factor influencing invertebrate abundance was structure, arrangement, and floristic diversity of the plant community. Finally, body weights of captured birds from within our study sites were well within the normal range reported by Brennan (1999) and were not significantly different on an annual or seasonal basis (Walsh 2004), also suggesting that food supply was not limiting for bobwhites.

We recommend management efforts in similar mixed shortleaf pine-oak forests aimed at increasing bobwhite densities include thinning to reduce midstory cover and frequent fire to maintain park-like



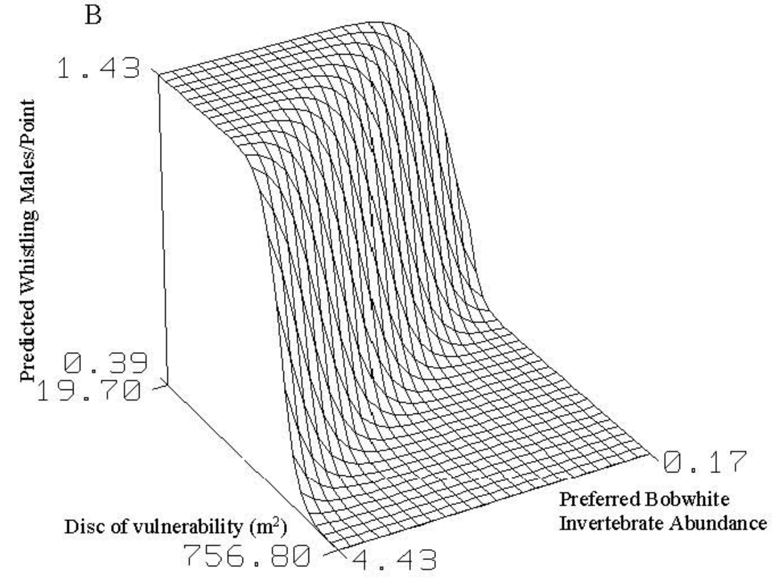


Figure 3: Artificial neural network predictions on the response of northern bobwhite whistling males to percent forb cover (%) and disc of vulnerability (m^2) (A), and to preferred bobwhite invertebrate abundance (mean invertebrates/sample) and disc of vulnerability (m^2) (B) on the Ouachita National Forest, Arkansas, 1999 and 2000.

conditions. Pine-grassland restoration efforts as described here created usable space (permanent understory woody cover in low basal area stands) for bobwhites (Cram et al. 2002). However, Walsh (2004) reported a winter shift in usable space in the same study area from treated stands as described here to thinned stands 2 years following fire and planted with shortleaf pine (regeneration stands). Planting food plots or providing supplemental feed on similar sites following thinning and fire would seem to be unnecessary based on the abundance of invertebrate and plant food items produced by thinning and fire. A final point is that our study area was managed toward ecosystem management goals on a landscape level not specifically for bobwhites. To reconcile the relative importance of the usable space vs. food quantity issue more work is needed.

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Winter Food Habits and Preferences of Northern Bobwhites in East Texas

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During late winter, 1994 and 1995, we investigated food habits and preferences of northern bobwhites (*Colinus virginianus*; hereafter, bobwhites) collected on forested lands in east Texas. Crops for bobwhites were collected from areas under 3 management regimes, namely intensively managed for bobwhites (QMA) (i.e., tree basal area reduced, annually burned, numerous multi-stage food plots, etc.), extensively managed for timber and wildlife (NBS) (i.e., burned every 3-5 years, scattered 2-stage food plots with corn feeders), and unmanaged for wildlife (i.e., burned every 5-7 years). With years pooled, partridge pea (*Cassia fasciculata*), Hercules club (*Zanthoxylum clava-herculis*), and pine (*Pinus* spp.) seeds, and clover leaflets (*Trifolium* spp.) comprised 93% by weight of foods of 79 bobwhites foods on QMA. On NBS, 81% of 40 bobwhite diets was butterfly pea (*Centrosema virginianum*), browntop millet, pine, wild bean (*Strophostyles* spp.), and corn seeds and clover leaflets; millet and corn were from food plots and feeders, respectively. For unmanaged areas, 79% of 19 bobwhite diets was butterfly pea, rush (*Juncus* spp.), pine, partridge pea, and American beautyberry (*Callicarpa americana*) seeds, and clover leaflets. Top-ranked food items on QMA were pine, hairy vetch, and Hercules club seeds in 1994 and butterfly pea, partridge pea, and wax myrtle (*Myrica cerifera*) seeds in 1995 ($P < 0.05$). On NBS, hawthorn (*Crataegus* spp.) and beautyberry seeds were top-ranked in 1994 as were kobe lespedeza, wild bean, and butterfly pea seeds in 1995. On unmanaged areas, butterfly pea and partridge pea seeds and clover leaflets were highest ranked in 1995. On forested lands, activities (e.g., disking, burning, establishing food plots) which provide seed-bearing plants, especially legumes, and clover greenery benefit bobwhites.

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Key words: *Colinus virginianus*, food availability, food habits, food preferences, Johnson's rank method, northern bobwhite, PREFER

Introduction

Northern bobwhite populations have been declining for at least 50 years (Brennan 1991, Church et al. 1993). Loss of habitat is the primary factor contributing to this decline (Goodrum and Reid 1954, Landers and Mueller 1986). Management practices targeted toward increasing bobwhite populations should include providing year round food for the species (Stoddard 1931); winter food supplies, however, are often scarce (Landers and Mueller 1986). Therefore, providing such foods is an important aspect of managing the species (Stoddard 1931, Lay 1965, Jackson et al. 1987).

Although biologists and land managers regularly plant food plots to meet winter food needs, literature on the subject is contradictory. Both Stoddard (1931) and Rosene (1969) pointed out successful managers who utilized food plots, yet both clearly stated that bobwhites prefer seeds of native plant species rather than those of introduced species. Robel et al. (1974) and Landers and Mueller (1986) suggested that food plots served as a safeguard against native food scarcities. They believed that bobwhites are opportunistic feeders and select what is most readily available. However, in central Florida, bobwhites selected slough-grass seeds (*Scleria muhlenbergii*) over those of wax myrtle (Laessle and Frye 1956). Slough-grass seeds had a much higher nutritional value than wax myrtle seeds, and since both were available, the authors reasoned that

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bobwhites selected items that best met their dietary needs.

Stoddard (1931) was the first biologist to study food habits of bobwhites. His studies and those of others (Lay 1965, Rosene 1969) were based on analyses of bobwhite crops from hunter-killed birds. Other studies examined food availability by collecting food items from the litter and surface soil (Haugen and Fitch 1955, Ault and Stormer 1983). However, no known studies have compared food utilization to food availability and thus developed food preferences of bobwhites. The objectives of this study were to compare winter food habits, availabilities, and preferences of bobwhites on 3 forested areas, each subjected to a different management regime. One area was intensively managed for bobwhites and another was extensively managed for wildlife and timber production. The third area was forested lands generally managed for timber production; for the purpose of this study, these lands will be referred to as unmanaged. All areas were similar in terrain, elevation, soil type, and timber type (Dietz 1999), but differed in the management objectives applied to them. Our null hypothesis was that bobwhites showed no preferences among foods consumed, regardless of management regime.

Study Areas

The Pineywoods of east Texas are generally comprised of pure pine and mixed pine-hardwood types. Soils are primarily alfisols, ultisols, and vertisols in the uplands (Gould 1962). In stands where bobwhites were collected, pines contributed >70% of both canopy cover and basal area. In natural stands, overstory pines were 50 to 60 years old and average basal area was about 22 m^2/ha. Dominant pine species were loblolly (*Pinus taeda*) and shortleaf (*P. echinata*), and dominant hardwood species were bitternut hickory (*Carya cordiformis*), black hickory (*C. texana*), post oak (*Quercus stellata*), southern red oak (*Q. falcata*), white oak (*Q. alba*), and sweetgum (*Liquidambar styraciflua*). Flowering dogwood (*Cornus florida*) was a common midstory tree.

The intensively managed study area, known as the Quail Management Area (QMA), consisted of 607 ha in the South Boggy Slough Hunting and Fishing Club. The club was in Trinity County, 16 km southwest of Lufkin, Texas. The QMA was established in 1989 by Temple-Inland Forest Products Corporation as a northern bobwhite research area. Parsons et al. (2000) described habitat modification on the QMA in detail.

There were 69 permanent food plots on the QMA, totaling 81 ha; size of the plots ranged 0.8-2.0 ha. Mean distance between plots was 296 m. Each plot contained a fall/winter section, a spring/summer section, and a border of 3- to 5-year-old rough. Larger plots had first-year fall/winter fallow and second-year fall/winter fallow sections. Seed mixes planted during the fall included crimson clover, red clover, winter wheat, hairy vetch, kobe lespedeza, and partridge pea. Spring seed mixes included Egyptian wheat, browntop millet, pearl millet, joint vetch, and cowpeas. Roadsides were planted to similar species (Dietz 1999).

The extensively managed study area was North Boggy Slough Hunting and Fishing Club (NBS). This club was approximately 13 km north of the QMA. It was managed for timber, white-tailed deer (*Odocoileus virginianus*), eastern wild turkeys (*Meleagris gallapavo*), and red-cockaded woodpeckers (*Picoides borealis*) (RCW). The habitat included 1,215 ha of hardwood bottoms and 2,025 ha of pine and mixed pine-hardwood uplands. In natural upland stands, basal areas ranged 21-28 m^2/ha. However, in RCW clusters and foraging areas, stands had been thinned to approximately 14 m^2/ha and the hardwood midstory removed. There were 50 permanent food plots on NBS, totaling 162 ha. Food plots on NBS had spring/summer and fall/winter sections, but most had no surrounding rough. Each food plot had a feeder which distributed 45 kg of corn per week during fall and winter. Seed mixes for food plots and roadsides were similar to those used on the QMA, but lacked partridge peas (Dietz 1999).

There were 3 primary differences in habitat management on the QMA and NBS. Nearly 15% of the QMA was planted annually whereas only 5% of NBS was so planted. Most of the QMA was on an annual burning cycle whereas NBS was on a 3- to 5-year cycle. Also, fallow disking took place in the woods on the QMA, but not on NBS.

Unmanaged areas were comprised of both private and United States Forest Service (USFS) lands. Forests on these large tracts (>500 ha) ranged from young pine plantations to mature mixed pine-hardwood stands; the tracts were primarily managed for pine sawlog and pulpwood production. Generally, little wildlife management had been implemented in these lands. The exception was the protection and management of RCW clusters. These cluster sites, most of which occurred on USFS lands, had been thinned to approximately 14 m^2/ha basal area and the hardwood midstory had been reduced or removed. Some stands were also on a 5- to 7-year burning cycle.

Methods

Bobwhites were harvested over pointing bird dogs during January and February 1994 and 1995. Upon harvest, all birds from a covey were placed in a bag marked with the covey identification number. Prior to leaving the area, the location where the covey was first contacted was flagged.

In the lab, each bird was weighed, sexed, and aged, then dissected and its crop and proventriculus removed. Contents of the crop and proventriculus were sorted, and each plant and animal food item identified to the lowest possible taxon using guides and keys by Landers and Johnson (1976), Rosene and Freeman (1988), and a personal seed collection. Once identified, each taxon was dried at 38° C for 48 hours and weighed to the nearest 0.0001 g using an electronic balance.

Food availability data were collected as soon as possible after each bird was bagged. If more than 1 bird was collected from a covey on the same day, only 1 food availability sample was gathered. On both the QMA and NBS, food available to each bobwhite (or covey) was sampled in a forested area (woods sample) and in 2 food plots. If the first contact with a covey occurred in the forest, the 2 food plots nearest the point of covey contact were sampled. If the first contact occurred in a food plot, the woods sample was initiated 30 m from the food plot edge which was nearest the point of covey contact.

In forested areas, food availability was sampled at 5 points (i.e., subsamples). The first subsample was taken at the point at which the covey was initially contacted. Each of the other 4 subsamples was collected 20 m from the first in a randomly selected direction. For each subsample, a 21.6-cm diameter ring was placed on the ground, then leaf litter, bark, and other large debris removed from within it. Next, the ground surface within the ring was vacuumed for 30 seconds using a hand-held, battery operated wet/dry vacuum (Worthington et al. 2004). Finally, herbaceous vegetation (i.e., greenery) within 15 cm of ground level was gathered. All potential food items, soil, debris, and green vegetation collected during the 5 subsamples were placed in a labeled paper bag.

Food plots were sampled in a manner similar to that of the woods samples. Each section of each food plot was sampled, but with only 2 subsamples. Potential food items gathered in each section of the 2 food plots were pooled (thus 4 subsamples per section), allowed to air dry, then temporarily stored in labeled plastic bags containing moth balls. Thereafter, potential food items were sorted, identified to the lowest possible taxon, and dried and weighed using the same methods as used for foods consumed.

Data Analyses

Thomas and Taylor (1990) referred to field studies that compared resource use and availability as selectivity studies. Their evaluation of study designs and tests for studies such as ours suggested the use of the Chi-square test of homogeneity, the Johnson rank method (i.e., the program PREFER), the Friedman test, or the Quade test. We chose the Johnson rank method (Johnson 1980) because it is less sensitive to the subjective inclusion or exclusion of resources (Alldredge and Ratti 1986, Thomas and Taylor 1990), does not require usage and availability to be estimated without error (Johnson 1980), ranks the order of individual components, and permits statistical comparisons. Because this method ranks the components, potential food items that are abundant but scarcely consumed are not dismissed as having little or no value (Johnson 1980). In this study, we included all consumed foods that comprised $\geq 1.0\%$ by weight. However, potential foods in availability samples not consumed by any bird were not tested. All statistical tests were made at $\alpha = 0.05$.

Results

During January - February 1994, 39 bobwhites were collected, 26, 11, and 2 from the QMA, NBS, and unmanaged lands, respectively. In 1995, 101 bobwhites were collected, 54 from the QMA, 29 from NBS, and 18 from unmanaged areas. However, 2 digestive tracts from 1995 were unusable, 1 from the QMA and the other from unmanaged areas. There were digestive tracts from 25, 19, 49, and 45 adult males, adult females, subadult males, and subadult females, respectively.

Foods Consumed

Digestive tracts of 138 bobwhites collected during the winters of 1994 and 1995 contained 78 identifiable plant foods. Sixty taxa were seeds and 18 were greenery; 12 taxa occurred as both seeds and greenery, thus 66 identifiable plant food items. Animal matter from 12 orders was also recorded. Excluding grit, bobwhites collected on the QMA consumed 35 and 54 different food items in 1994 and 1995, respectively (Dietz 1999, pgs 35-39). Twenty-six and 55 identifiable food items were consumed in 1994 and 1995, respectively, by birds collected on NBS. Birds collected on unmanaged areas yielded 8 different foods in 1994 and 37 in 1995. However, sample size was only 2 birds in 1994.

1994 - Pine seeds made up the major portion (56%) of bobwhite diets in 1994. On the QMA, pine, partridge pea, and Hercules club seeds and clover leaflets comprised the bulk (93%) of foods consumed (Table 1). Partridge pea was the only planted species with seeds that comprised >10% of the digestive tract weight. Leaflets from planted clovers comprised 91% of greenery weight; no other taxa of greenery contributed $\geq$1% to the total weight. Beetles and butterfly (Lepidoptera) pupae each occurred in approximately 15% of the digestive tracts, but contributed <1% to the digestive tract weight (Dietz 1999). For bobwhites on NBS, seeds of pine, planted browntop millet, and corn from feeders made up 87% of the total digestive tract weight. Planted clovers contributed most of the greenery (Table 1). Butterfly pupae and snails occurred in 36 and 27% of the digestive tracts, respectively, but comprised <1% of the total weight. The 2 birds from unmanaged areas consumed American beautyberry and pine seeds (93%) and greenery of yaupon (*Ilex vomitoria*) and American beautyberry (Dietz 1999).

1995 - In the second winter, pine seeds made up a much smaller portion (9%) of bobwhite diets. Seeds of native wax myrtle, butterfly pea, and pine and planted partridge pea and kobe lespedeza comprised approximately 59% of QMA bobwhite diets. The most used food item was leaflets from planted clovers which comprised 26% of the total weight, 96% of the greenery weight, and occurred in 50 of the 53 birds (Table 1). Animal matter consumed by bobwhites included beetles (Coleoptera), true bugs (Hemiptera), and ants (Hymenoptera) (Dietz 1999). Important food items on NBS were seeds of native wild bean, butterfly pea, pine, partridge pea, and planted browntop millet and kobe lespedeza, and clover greenery (Table 1). Although animal matter made up a very small part of what NBS bobwhites ate (Table 1), beetles and snails each occurred in $\geq$20% of the digestive tracts (Dietz 1999). For bobwhites from unmanaged areas, butterfly pea seeds made up the greatest total weight and occurred in 41% of digestive tracts. Partridge pea, pine, rush, and snake root (*Psoralea psoralioides*) seeds and greenery from wild clover were also important food items (Table 1). Beetles and butterfly pupae were the most frequently consumed animal foods, but both occurred in small amounts (Dietz 1999).

Table 1: Number of northern bobwhites and mean percent composition by weight of planted and wild food items in digestive tracts of 138 bobwhites collected in east Texas, winters 1994 and 1995, from areas intensively (QMA), extensively (NBS), and unmanaged for wildlife. Species planted in food plots on the QMA and NBS are indicated by PI and PE, respectively. Values < 0.05% are shown as trace (tr.).

	1994			1995			
Taxon	QMA ($n = 26$) Count (%)	NBS ($n = 11$) Count (%)	Unmanaged ($n = 2$) Count (%)	QMA ($n = 53$) Count (%)	NBS ($n = 29$) Count (%)	Unmanaged ($n = 17$) Count (%)	Pooled ($n = 138$) Count (%)
Seeds							
Partridge pea (PI)	7 (13.4)			19 (24.9)	6 (5.8)	5 (11.5)	37 (15.3)
Butterfly pea		1 (0.1)		20 (4.6)	10 (9.5)	7 (27.3)	38 (4.4)
Lespedeza (PI, PE)	2 (tr.)			13 (5.6)	16 (5.8)	3 (1.1)	34 (2.9)
Wax myrtle				11 (13.0)			11 (3.9)
Browntop millet (PI, PE)		2 (16.6)			3 (15.4)		5 (4.5)
Pine	21 (61.7)	8 (43.4)	2 (57.1)	16 (11.2)	12 (7.0)	6 (7.5)	65 (30.6)
Wild bean				8 (1.7)	9 (24.7)	5 (3.5)	22 (5.5)
Hairy vetch (PI, PE)	6 (2.7)	1 (0.1)		1 (tr.)		2 (1.0)	10 (1.2)
Hercules club	11 (10.4)	2 (0.5)		11 (2.3)	9 (2.6)	2 (0.2)	35 (4.5)
Corn	1 (tr.)	3 (26.8)		3 (1.3)	1 (0.7)		8 (3.0)
Others (n=50)	(1.8)	(3.9)	(39.9)	(6.7)	(4.0)	(21.9)	(5.6)
Subtotal	(90.0)	(91.4)	(97.0)	(71.3)	(75.5)	(74.0)	(81.4)
Greenery							
Clovers (PI, PE)	23 (7.9)	3 (0.2)		50 (26.2)	26 (21.1)	13 (21.7)	115 (15.3)
Others (n=17)	(0.7)	(2.9)	(2.0)	(1.1)	(1.6)	(2.1)	(1.3)
Subtotal	(8.6)	(3.1)	(2.0)	(27.3)	(22.7)	(23.8)	(16.6)
Animal (n=12)	(1.0)	(1.9)	(0.1)	(0.8)	(1.0)	(1.5)	(1.0)
Grit (Rock and Shot)	4 (0.4)	5 (3.6)	1 (0.9)	9 (0.6)	8 (0.8)	2 (0.7)	(1.0)
Total	(100.0)	(100.0)	(100.0)	(100.0)	(100.0)	(100.0)	(100.0)
Avg. Weight of Digestive Tract Contents (g)	1.8465	2.0305	1.0477	1.3905	1.6396	0.5963	1.4715

Table 2: Number of samples and mean percent composition by weight of food items available to northern bobwhites on areas intensively (QMA), extensively (NBS), and unmanaged for wildlife during the winters of 1994 and 1995. Species planted in food plots on the QMA and NBS are indicated by PI and PE, respectively. Values < 0.05% are shown as trace (tr.).

	1994			1995			
Taxon	QMA ($n = 12$) Count (%)	NBS ($n = 7$) Count (%)	Unmanaged ($n = 1$) Count (%)	QMA ($n = 20$) Count (%)	NBS ($n = 12$) Count (%)	Unmanaged ($n = 6$) Count (%)	Pooled ($n = 58$) Count (%)
Seeds							
Wooly croton	10 (4.7)	2 (0.7)		4 (0.1)	2 (0.1)	3 (0.6)	21 (1.3)
Browntop millet (PI, PE)	9 (0.4)	3 (0.2)		20 (3.3)	11 (6.3)		43 (2.5)
Pine	11 (1.2)	6 (0.8)	1 (9.9)	17 (0.7)	9 (1.0)	4 (1.9)	48 (1.0)
Oak		1 (3.2)		3 (1.2)	2 (3.7)		6 (1.7)
Egyptian wheat (PI, PE)	10 (9.2)	2 (1.0)		16 (1.3)	1 (0.2)		29 (2.9)
Others (n=52)	(3.9)	(0.9)	(12.7)	(5.4)	(4.9)	(4.1)	(4.0)
Subtotal	(19.4)	(6.8)	(22.6)	(12.0)	(16.2)	(6.6)	(13.4)
Greenery							
Pigweed	9 (2.3)	6 (1.9)	1 (4.0)	16 (1.2)	8 (1.7)	3 (0.6)	43 (1.6)
Brome grass	4 (0.9)	3 (0.4)		4 (1.9)		2 (32.5)	13 (2.4)
Ryegrass (PI, PE)	6 (1.8)	1 (1.7)		11 (4.9)	4 (0.3)		22 (2.5)
Panic grass	9 (2.9)	2 (0.3)	1 (67.8)	13 (8.1)	4 (1.3)	4 (47.2)	33 (6.3)
Blackberry	11 (10.3)	2 (1.1)		14 (3.2)	5 (0.3)	2 (1.6)	34 (3.9)
Nightshade	4 (0.9)	2 (0.2)		13 (2.2)	6 (0.8)	2 (0.6)	27 (1.2)
Clovers (PI, PE)	12 (23.4)	7 (38.9)	1 (1.6)	20 (24.7)	12 (33.6)	2 (4.1)	54 (27.5)
Winter wheat (PI, PE)	12 (29.3)	7 (39.2)		19 (33.9)	11 (41.5)		49 (33.6)
Hairy vetch (PI, PE)	12 (4.4)	5 (6.8)		14 (2.5)	3 (0.3)	1 (1.2)	35 (3.3)
Others (n=42)	(4.4)	(2.7)	(4.0)	(5.3)	(4.0)	(5.6)	(1.3)
Subtotal	(80.6)	(93.2)	(77.4)	(87.9)	(83.8)	(93.4)	(86.6)
Animal (n=3)	(tr.)			(0.1)			(tr.)
Total	(100.0)	(100.0)	(100.0)	(100.0)	(100.0)	(100.0)	(100.0)
Avg. wt of sample	5.8714	7.4760	1.3672	5.3310	4.8484	2.2696	5.2191

Table 3: Mean differences and groups of preference rankings of foods consumed by northern bobwhites on areas intensively (QMA) and extensively (NBS) managed for wildlife in east Texas, 1994. Groups with the same capital letters are not different ($P > 0.05$). Seeds and greenery are referenced as s and g, respectively.

Management Regime	Food	Difference	Group
QMA^a	Pine (s)	4.27	A
	Hairy vetch (s)	3.79	A
	Hercules club (s)	2.98	AB
	Partridge pea (s)	2.6	BC
	Hawthorn (s)	2.42	BC
	Rye grass (g)	0.98	C
	Clover (g)	-1.19	D
	Hairy vetch (g)	-1.77	D
	Panic grass (g)	-2.33	DE
	Wooly croton (s)	-3.52	E
	Egyptian wheat (s)	-4.13	E
	Winter wheat (s)	-6.73	F
NBS^a	Hawthorn (s)	1.18	A
	American beautyberry (g)	1.14	A
	Corn (s)	0.77	AB
	Pine (s)	0.68	AB
	Oak (s)	0.64	AB
	Browntop millet (s)	0.05	AB
	Hairy vetch (g)	-1.05	BC
	Clover (g)	-3.41	C

[a] Critical value for Waller-Duncan: QMA = 1.92, NBS = 2.87

Foods Available

Food availability samples were collected for each covey contact. On the QMA, 12 and 20 samples were gathered for 26 and 54 birds collected in 1994 and 1995, respectively. On NBS, 7 availability samples gathered in 1994 and 12 in 1995 represented the potential foods for 40 birds collected there. For 20 birds collected from the unmanaged areas (2 in 1994 and 18 in 1995), 7 availability samples were gathered, 1 in 1994 and 6 in 1995.

1994 - As expected, greenery from species planted in food plots dominated food availability weights on the QMA and NBS; greenery from planted clovers and winter wheat exceeded 50% of the weight on each area each year. On the QMA, seeds of native wooly croton (*Croton capitatus*), pine, and nightshade (*Solanum* spp.) and planted Egyptian wheat comprised 15% of the weight of foods available. Browntop millet seeds occurred in 75% of samples but contributed little to the total weight of potential food items. Greenery from native pigweed (*Amaranthus* spp.), panic grass (*Panicum* spp.), and blackberry (*Rubus* spp.) comprised 16% of the weight and each occurred in ≥75% of the samples (Table 2). Oak mast comprised the greatest weight of seeds on NBS, but occurred in only a single sample. Pine, panic grass, and paspalum (*Paspalum* spp.) seeds occurred in most samples but contributed little weight (Dietz 1999). Native greenery made up <5% of total food availability weight. On unmanaged lands, pine, smartweed (*Polygonum* spp.), and

Table 4: Mean differences and groups of preference rankings of foods consumed by northern bobwhites on areas intensively (QMA), extensively (NBS), and unmanaged for wildlife in east Texas, 1995. Groups with the same capital letters are not different ($P > 0.05$). Seeds and greenery are referenced as s and g, respectively.

Management Regime	Food	Difference	Group
QMA^a	Butterfly pea (s)	4.72	A
	Partridge pea (s)	4.11	A
	Wax myrtle (s)	3.86	A
	Lespedeza (s)	2.07	BC
	Oak (s)	1.35	CD
	Hercules club (s)	1.24	CD
	Ryegrass (g)	0.61	DE
	Pine (s)	-0.23	DE
	Clover (g)	-0.27	E
	Hairy vetch (g)	-0.9	EF
	Beggarweed (s)	-1.38	F
	Egyptian wheat (s)	-2.38	F
	Panic grass (s)	-4.57	G
	Winter wheat (g)	-6.86	H
NBS^a	Lespedeza (s)	3.68	A
	Wild bean (s)	3.65	A
	Butterfly pea (s)	2.93	AB
	Partridge pea (s)	1.94	BC
	Hercules club (s)	1.41	BCD
	Ryegrass (g)	0.37	CDE
	Oak (s)	-0.13	DEF
	Pine (s)	-0.26	DEF
	Dogwood (s)	-0.33	EF
	Carolina jessamine (g)	-0.41	EF
	Egyptian wheat (g)	-0.7	EF
	Panic grass (s)	-1.56	F
	Browntop millet (s)	-3.98	G
	Winter wheat (g)	-6.61	H
$Unmanaged^a$	Butterfly pea (s)	2.25	A
	Clover (g)	2.18	A
	Partridge pea (s)	2.07	A
	Yaupon (g)	0.71	AB
	Rush (s)	0.61	AB
	Snake root (s)	0.5	AB
	Carolina jessamine (g)	-1.25	BC
	Pine (s)	-1.32	BC
	Panic grass (s)	-1.79	C
	Panic grass (g)	-3.96	C

[a] Critical values for Waller-Duncan: QMA = 1.79, NBS = 1.94, Unmanaged = 3.05

wild sorghum (*Sorghum* spp.) comprised the bulk of seeds in the single sample. Pigweed, yaupon, panic grass, and wild clovers provided most greenery (Table 2).

1995 - On the QMA, seeds made up 12% of the available foods. Only acorns and seeds of native beggarweed (*Desmodium* spp.) and Hercules club and planted Egyptian wheat and browntop millet comprised weights $\geq$1%. Native greenery with weights $\geq$1% were blackberry and panic grass. On NBS, seeds made up approximately 16% of the total weight of potential foods (Table 2). Browntop millet and acorns comprised the majority of the seed weight; acorns occurred in only 2 samples, however. Seeds of wild panics, beggarweed, and smartweed were relatively common but made up small proportions of available foods (Dietz 1999). Native greenery from panic grass, pigweed, nightshade, and chickweed (*Stellaria media*) each made up a small part of the weight, but each occurred in $\geq$33% of samples. On unmanaged areas, pine seeds made up the greatest biomass and were the most available food item (Table 2). Panic grass and wooly croton seeds also were common, but contributed little to the total weight. Greenery from panic grass, brome grass (*Bromus* spp.), Carolina jessamine (*Gelsemium sempervirens*), and wild clover was also relatively common (Table 2, Dietz 1999).

Food Preferences

In 1994, bobwhites on the QMA selected pine, hairy vetch, and Hercules club seeds over all other foods ($P < 0.05$). Partridge pea and hawthorn seeds ranked second in importance. Seeds of panic grass and Egyptian wheat and greenery of winter wheat were ranked lowest (Table 3). Highly ranked seeds on the NBS in 1994 were hawthorn, pine, oak, browntop millet, and corn from feeders. American beautyberry was the green vegetation selected; leaves of hairy vetch and clover were rated lowest (Table 3). As only 1 food availability sample was collected on an unmanaged area, preference was not tested.

In 1995, seeds selected by bobwhites on the QMA were butterfly pea, partridge peas, and wax myrtle. Seeds of kobe lespedeza, Hercules club, oak, pine, and beggarweed and greenery of planted ryegrass, hairy vetch, and clover were centrally ranked, while seeds and greenery of other planted species ranked lowest. For NBS bobwhites, seeds of planted kobe lespedeza and 4 native species were in the 2 highest ranked groups. Seeds of 3 native tree species and greenery from 2 planted and a native species were centrally ranked. Seeds of planted browntop millet and greenery of winter wheat were included in the lowest groups. Bobwhites from unmanaged areas selected butterfly pea, partridge pea, rush, and snake root seeds and clover greenery. Pine seeds were ranked mid-range, and panic seeds and panic greenery ranked lowest (Table 4).

Discussion

Foods Consumed

Pine seeds and clover greenery were the foods most utilized by bobwhites in this study. Stoddard (1931) found that pine seeds constituted 32.5% of bobwhite diets in the winter months of 1924, but only 4.2% for the same months the following years of a 5-year study. He pointed out that 1924 was an exceptional year for pine mast production. Likewise, during this study, there apparently was an exceptional pine seed crop in east Texas in the fall of 1993. Although this was not reflected in weights from 1994 availability samples, it was evident in the frequencies; pine seeds were recorded in 18 of the 20 samples.

Although equally available each year, clover greenery on the QMA and NBS was consumed in greater quantities and more frequently in 1995 than in 1994. It was also an important food component for bobwhites on unmanaged areas in 1995. Landers and Mueller (1986) wrote that planted foods increase in importance when native foods become scare. If so, bobwhites may have utilized clover in 1995 in the absence of pine seeds. Other researchers have reported similar shifts (Stoddard 1931, Reid and Goodrum 1959).

Seeds of partridge pea were consumed on the

QMA at much higher rates than on NBS or unmanaged areas. It was planted and grew wild on the QMA and grew wild on NBS and unmanaged areas. In the absence of an abundant supply of partridge pea seeds, NBS bobwhite focused on seeds of wild bean, butterfly pea, and brown millet; although browntop millet seeds were equally available on the QMA, they were not consumed there. On unmanaged areas, bobwhite did not benefit from planted foods, thus consumed available native foods, including seeds of butterfly pea, wild partridge pea, and wild bean.

Foods Available

On the QMA, food availability samples contained fewer taxa of seeds (44) than greenery (48). The relationship was reversed on the other 2 areas, with 35 and 23 taxa of seeds and greenery on NBS and 20 and 12 on unmanaged areas. Although unequal numbers of availability samples may have contributed to these differences, the multi-sectional make up of food plots on the QMA was probably most important. Stoddard (1931), Lay (1965), and Rosene and Freeman (1988) reported that bobwhites prefer the first and second stages of succession and that disking stimulates the abundance and diversity of weed species. Buckner and Landers (1979) found that ground which lay fallow for up to 2 years after disking had more herbaceous weeds than did ground which was untouched or burned. These findings suggest that 1- and 2-year fallow sections of food plots on the QMA increased the number of herbaceous species.

Food Preferences

On both the QMA and NBS, consumption rates and preference rankings of pine seeds were much higher in 1994 than in 1995. Subjectively, pine seed were much more abundant in fall 1993 than in fall 1994. However, the sampling technique we used showed little difference in abundances between years. It is unclear why, but pine seeds may have been localized under and around specific trees and missed by the sampling technique. Alternatively, pine seeds are an important food to many species and may have been consumed as soon as they fell. Regardless, the combination of high consumption and relatively low apparent availability resulted in a high preference ranking in 1994. Conversely, moderate consumption and low apparent availability resulted in a moderate preference ranking in 1995.

In the absence of an abundant pine mast crop in fall 1994, bobwhites shifted food preferences to native and planted legumes in winter 1995. Researchers have long recognized the importance of legumes as a food source for bobwhites (Stoddard 1931, Lay 1965, Peoples et al. 1994). In an investigation of 1,400 bobwhite digestive tracts, 12 of the 14 most frequently consumed foods were legumes; greenery and pine seeds were the remaining 2 items (Rosene 1969). Stoddard (1931) noted that partridge peas occurred in ≥70% of the digestive tracts of nearly 3,000 bobwhites. Lay (1965) recognized partridge pea as the most important wild legume and kobe lespedeza as the most important planted legume in east Texas.

Greenery from species planted on food plots on the QMA and NBS generally ranked in the mid-to-low ranges of preference. Such greenery was readily available, thus its low rankings are not surprising. However, the high ranking of clover on unmanaged areas demonstrates the importance of greenery to bobwhites. Baldwin and Handley (1946) found clover greenery in 149 of 495 bobwhite digestive tracts and Landers and Mueller (1986) reported greenery as the most frequently consumed food in early spring. Our results suggest that in the absence of greenery in food plots, bobwhites utilized greenery from wild clover if available, or yaupon, American beautyberry, and other species if not.

Management Implications

This study suggested that food plots benefit bobwhites in forested areas of east Texas. The birds readily consumed seeds of planted partridge pea, lespedeza, and hairy vetch and greenery of planted clover. Goodrum and Reid (1954) suggested planting legumes to supplement native food supplies for bobwhites in eastern Texas and western Louisiana. They pointed out that populations in forested areas were limited by often unpredictable native food supplies. The unpredictable nature of native foods and the value of foods from planted species were clearly demonstrated in this study wherein diets of birds from the QMA and NBS were dominated by pine seeds in 1994 and by seeds and greenery of planted species in 1995. Managers concerned with bobwhite populations should consider supplemental food plots which include partridge pea, kobe lespedeza, hairy vetch, and clover for greenery.

Other species that may warrant planting, as indicated by this study, are browntop millet and Egyptian wheat. In the absence of partridge peas, NBS birds consumed relatively high proportions of browntop millet each year. Planted on fertile, well-drained soils, Egyptian wheat will exceed 2 m in height. It offers value as a nurse crop to planted legumes and the strong stalks easily supported twining vines of hairy vetch. It also provides cover for bobwhite hens with broods.

Finally, improvement harvests which open canopies in pine stands encourage seed production by overstory trees and increase accessibility to seeds on the ground by bobwhites. In east Texas, however, such harvests must be coupled with regular prescribed burning or disking to prevent encroachment by woody species.

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Habitat Use and Survival of Gray Partridge Pairs in Bavaria, Germany

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Gray partridge (*Perdix perdix*) habitat studies have been undertaken in a number of countries but have generally focused on winter and brood rearing. We monitored survival of grey partridge pairs relative to habitat during the breeding season. Our study area was located near Feuchtwangen in north-west Bavaria, Germany. During 1991 to 1994, we used compositional analysis to assess habitat with survival and year as covariates for 38 radio-tagged partridge pairs. Comparing study area habitat to habitats within pair home ranges, we found overall habitat use was non-random with no year effect but a significant effect of survival status. Stubble habitat ranked high for both survival categories, whereas those pairs where the radio-tagged bird died were more associated with meadow habitat. Comparing home ranges to individual radio locations, only surviving partridge used habitat differently from availability. Edge and set aside ranked high whereas meadow ranked low in usage. Our results suggest differences between habitats of partridge which died versus those that survived during breeding season. At the landscape level, association with meadow habitat suggests that it may provide cover but may also support predators. Within home ranges, we see edge and set aside possibly providing more cover diversity, suggesting predation avoidance for those that survived. Our data suggests that late winter and early spring periods, where survival may impact numbers of adults going into the breeding season and ultimately recruitment, are also crucial.

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Key words: agriculture, farming, habitat, pairs, survival

Introduction

Long-term decreases in numbers of gray partridge, *Perdix perdix*, in the hunting districts around Feuchtwangen, North Bavaria, Germany motivated the "Hunting Society of Feuchwangen" to carry out a habitat management program for gray partridge since 1985. Gray partridge habitat studies have been undertaken in a number of countries but have generally focused on winter and brood rearing (Potts 1986, Carroll 1990, Carroll et al. 1995, Church and Porter 1990, Kaiser 1998). This period is considered critical as winter family groups and coveys break up and pairs are formed; dispersing to nesting areas (Potts 1986). Subsequent recruitment will then depend in part on dispersal and survival of those breeding pairs (Potts 1986). Previously, Smith et al. (1982) and Church and Porter (1990) evaluated habitat use by breeding pairs.

In addition, there have been few studies linking habitat and individual survival. For example, Panek (1990) found in Poland during winter that coveys with home ranges closer to forests had higher mortality rates. However, in Bavaria, Kaiser (1998) found no relationship between habitat use and survival among winter coveys.

As part of a larger study to assess the effects of different types of habitat management on partridge populations and ecology, we monitored survival of grey partridge pairs relative to habitat during the breeding season.

Study Area

Our study area was located near Feuchtwangen (District Ansbach) in north-west Bavaria, 70 km

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Table 1: Log-ratio differences and rankings of individual habitat comparing 2^{nd} order habitat use (study area versus home range) for surviving partridge pairs ($n = 26$) during breeding season in Bavaria, Germany. For rankings, a larger number means that the habitat was most selected.

		sa	**st**	**ed**	**wc**	**ra**	**pf**	**me**	**rank**
set-aside	mean		-0.199	0.67	0.374	1.124	0.251	1.268	**5**
	SE		0.698	0.321	0.378	0.477	0.396	0.397	
	p		0.781	0.057	0.349	0.048	0.529	0.011	
stubble	mean	0.199		0.646	0.827	1.115	0.92	1.107	**6**
	SE	0.698		0.386	0.393	0.516	0.374	0.389	
	p	0.781		0.12	0.061	0.028	0.028	0.018	
edge	mean	-0.67	-0.646		-0.015	0.189	-0.02	0.426	**2**
	SE	0.321	0.386		0.235	0.274	0.253	0.212	
	p	0.057	0.12		0.944	0.501	0.928	0.048	
winter cereal	mean	-0.374	-0.827	0.015		0.209	-0.005	0.442	**3**
	SE	0.378	0.393	0.235		0.343	0.17	0.267	
	p	0.349	0.061	0.944		0.565	0.974	0.083	
rape	mean	-1.124	-1.115	-0.189	-0.209		-0.321	0.245	**1**
	SE	0.477	0.516	0.274	0.343		0.361	0.336	
	p	**0.048**	**0.028**	0.501	0.565		0.387	0.496	
plowed field	mean	-0.251	-0.92	0.02	0.005	0.321		0.447	**4**
	SE	0.396	0.374	0.253	0.17	0.361		0.28	
	p	0.529	**0.028**	0.928	0.974	0.387		0.135	
meadow	mean	-1.268	-1.107	-0.426	-0.442	-0.245	-0.447		**0**
	SE	0.397	0.389	0.212	0.267	0.336	0.28		
	p	**0.011**	**0.018**	**0.048**	0.083	0.496	0.135		

south-west of Nurnberg, Germany. It was situated between 427 m and 514 m above sea level. Of the total area in the district of Ansbach, 29% was covered by small woodlands and 60% was private farmland. Main agricultural crops were winter cereals (40%), maize, rape, and root crops (21%). About 39% of farmland consisted of permanent grassland (Reider 1984). Average field size was 1.5 ha and the length of permanent cover along ways, hedges and ditches was 18 km/km^2. Spring densities of partridge were 4-8 pairs/km^2 during 1992-1994 (Kaiser 1998).

Methods

During 1991 to 1994, 136 partridges were captured in autumn using mist nets (5 m x 18 m, mesh-width 3 x 3 cm). The birds were equipped with 7 g necklace radio tags (TW-3, Biotrack Co., UK) with a life expectancy of 7-8 months and a range of 800 - 1000 m. From capture to pairing, partridge were tracked using a Televilt RX-81 receiver and a two-element Yagi aerial. Partridge were located daily, but to avoid bias due to time of day, radiolocations were sampled throughout the day. We used "homing" techniques to verify individual locations. This allowed us to accurately place each location in one of the following habitat types: set-aside (SA; self-regenerated), cereal stubble (CS; also maize stubble), edge (ED; hedges, ways, ditches, field boundaries), oilseed-rape (RA), ploughed field (PF), permanent grassland (PG), and winter cereal (WC).

We used compositional analysis (Aebischer et al. 1993) to assess habitat with survival and year as covariates. We defined the breeding or spring season to encompass those dates during break up of win-

Table 2: Log-ratio differences and rankings of individual habitat comparing 2^{nd} order habitat use (study area versus home range) for partridge pairs ($n = 12$) that died during the breeding season in Bavaria, Germany. For rankings a larger number means that the habitat was most selected.

		sa	**st**	**ed**	**wc**	**ra**	**pf**	**me**	**Rank**
setaside	mean		-1.951	-0.368	-0.811	-1.384	-1.29	-0.915	**6**
	SE		0.318	0.682	0.573	1.537	0.701	0.704	
	p		0.033	0.594	0.212	0.527	0.103	0.236	
stubble	mean	1.951		1.609	1.473	0.608	1.036	1.19	**5**
	SE	0.318		0.58	0.313	1.306	0.399	0.504	
	p	**0.033**		0.043	0.015	0.73	0.033	0.053	
edge	mean	0.368	-1.609		0.27	-0.053	-0.136	-0.348	**3**
	SE	0.682	0.58		0.39	0.704	0.367	0.102	
	p	0.594	**0.043**		0.524	0.913	0.71	0.008	
winter cereal	mean	0.811	-1.473	-0.27		-0.865	-0.445	-0.636	**2**
	SE	0.573	0.313	0.39		0.636	0.095	0.344	
	p	0.212	**0.015**	0.524		0.21	0.001	0.08	
rape	mean	1.384	-0.608	0.053	0.865		0.474	-0.224	**4**
	SE	1.537	1.306	0.704	0.636		0.513	0.685	
	p	0.527	0.73	0.913	0.21		0.429	0.713	
plowed field	mean	1.29	-1.036	0.136	0.445	-0.474		-0.211	**0**
	SE	0.701	0.399	0.367	0.095	0.513		0.319	
	p	0.103	**0.033**	0.71	**0.001**	0.429		0.607	
meadow	mean	0.915	-1.19	0.348	0.636	0.224	0.211		**1**
	SE	0.704	0.504	0.102	0.344	0.685	0.319		
	p	0.236	0.053	**0.008**	0.08	0.713	0.607		

ter coveys to nesting. We analyzed habitat at two scales encompassing Johnson's (1980) 2^{nd} and 3^{rd} orders. For 2^{nd} order analysis we compared the proportions of habitats for the whole study area with those within convex polygon home ranges. For 3^{rd} order analysis we compared proportions of habitats within convex polygon home ranges to individual radio locations.

Results

We monitored survival status and habitat of 38 grey partridge pairs during 1992-1994 (Figure 1a). Comparing study area habitat to habitats within pair home ranges, we found overall habitat use was non-random ($\lambda = 0.4234, n = 38, P < 0.001$). We found no year effect ($\lambda = 0.629, P = 0.14$), but a significant effect of survival status ($\lambda = 0.5855, P = 0.005$) on habitat use (Figure 1b). Habitat proportions within home ranges compared to individual locations were different ($\lambda = 0.3820, P = 0.008$). We found year effect to be close to significant ($P = 0.05$) and a survival effect ($P = 0.01$). Comparing study area to home ranges, both surviving and non-surviving partridge used habitat differently from availability ($\lambda = 0.4388, n = 26, P = 0.005$ and $\lambda = 0.0769, n = 12, P = 0.001$, respectively). At this level, stubble ranked high for both survival groups, whereas those pairs where the radio-tagged bird survived were more associated with set aside and those that died more associated with meadow habitat (Tables 1 and 2). For both survival groups cereal, edge, and rape, ranked low.

Comparing home ranges to individual radio locations, only surviving partridge used habitat dif-

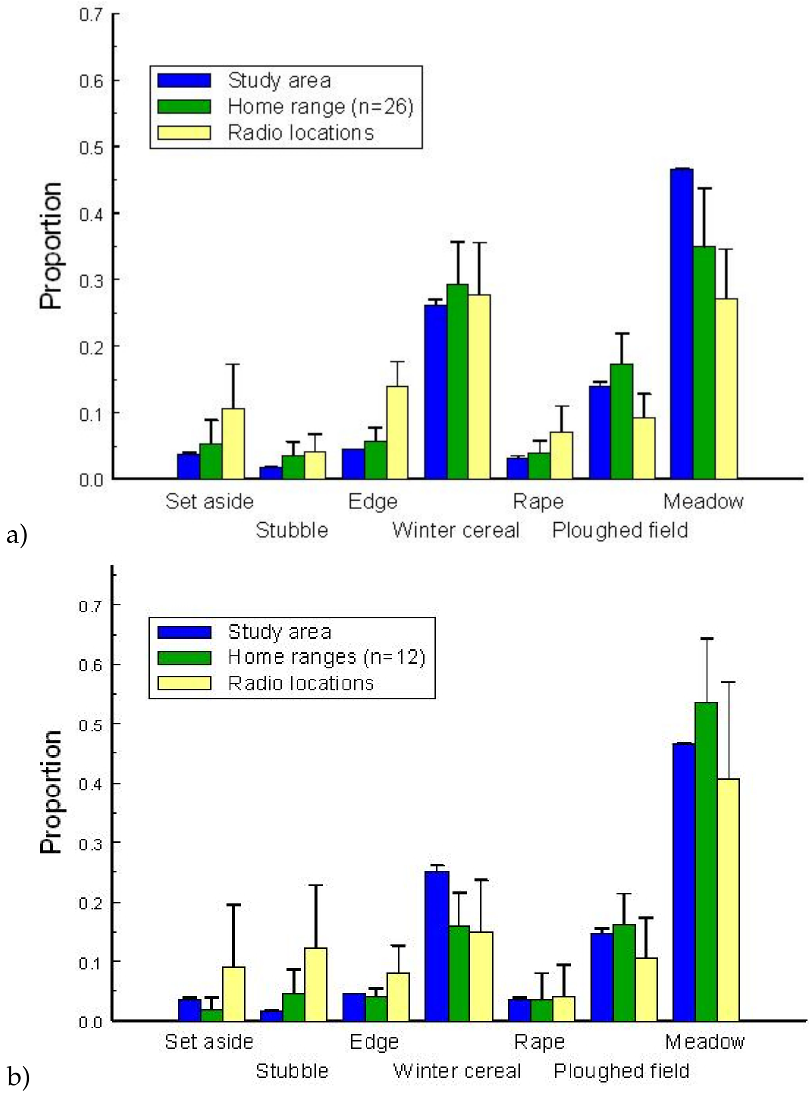

Figure 1: Habitat proportions (+SE) available on the study area, within convex polygon home ranges, and for individual radio locations of surviving gray partridge pairs (a) and those that died (b) during breeding season in Bavaria, Germany.

ferently from availability ($\lambda = 0.2004, n = 26, P < 0.001$, and $\lambda = 0.2632, n = 12, P = 0.69$, respectively). At this level of analysis (habitat within home ranges to individual radio locations), we found that edge and set aside ranked high, whereas meadow and cereal ranked low for surviving pairs (Table 3).

Discussion

Habitat associations relative to survival of gray partridge have been demonstrated during winter (Panek 1990, Carroll et al. 1995) and during brood rearing (Potts 1986). Our habitat results were similar to Church and Porter (1990) who found pairs selecting for grain stubble due to waste grain, which is found in abundance in these habitats. They also found that idle upland habitat, dominated by old

Table 3: Log-ratio differences and rankings of individual habitat comparing 3^{rd} order habitat use (home range versus radio locations) for surviving partridge pairs ($n = 26$) during breeding season in Bavaria, Germany. For rankings a larger number means that the habitat was most selected.

		sa	**st**	**ed**	**wc**	**ra**	**pf**	**me**	**rank**
setaside	mean		0.774	-0.024	1.283	0.098	2.045	1.323	**5**
	SE		0.469	0.401	0.191	0.606	0.297	0.241	
	p		0.142	0.951	0.001	0.873	0.002	0.002	
stubble	mean	-0.774		-0.952	0.446	-0.369	1.366	0.524	**3**
	SE	0.469		0.29	0.298	0.629	0.303	0.291	
	p	0.142		0.005	0.148	0.54	0.003	0.1	
edge	mean	0.024	0.952		1.086	0.251	1.545	1.111	**6**
	SE	0.401	0.29		0.301	0.412	0.288	0.277	
	p	0.951	**0.005**		0.002	0.548	0.001	0.001	
winter cereal	mean	-1.283	-0.446	-1.086		-1.168	0.44	0.015	**2**
	SE	0.191	0.298	0.301		0.394	0.248	0.2	
	p	**0.001**	0.148	**0.002**		0.019	0.096	0.944	
rape	mean	-0.098	0.369	-0.251	1.168		1.437	0.97	**4**
	SE	0.606	0.629	0.412	0.394		0.396	0.353	
	p	0.873	0.54	0.548	**0.019**		0.003	0.017	
plowed field	mean	-2.045	-1.366	-1.545	-0.44	-1.437		-0.514	**0**
	SE	0.297	0.303	0.288	0.248	0.396		0.203	
	p	**0.002**	**0.003**	**0.001**	0.096	**0.003**		0.016	
meadow	mean	-1.323	-0.524	-1.111	-0.015	-0.97	0.514		**1**
	SE	0.241	0.291	0.277	0.2	0.353	0.203		
	p	**0.002**	0.1	**0.001**	0.944	**0.017**	**0.016**		

field herbaceous cover, was selected preferentially. Smith et al. (1982) found that what they termed "idle" habitat was also preferred by pairs. However, on their study area in South Dakota, these habitats were usually grassy edges along crop fields, roads, and shelterbelts. These studies in both Europe and North America all suggest that habitat use during breeding season is driven by proximity to nesting sites. In both of those studies (Smith et al. 1982, Church and Porter 1990), home range and radio-location data were not divided. Our finding of greater selection of edge habitats and set aside among surviving pairs suggests that this trend is similar on our study area.

Our analysis of survival relative to habitat of breeding pairs suggests that there were significant differences between habitats of breeding individuals which died versus those that survived during breeding season. These differences are likely a function of landscape and local effects of predator distribution relative to land use. The association between partridges that died and meadow habitat suggests that this permanent habitat likely provides some cover and is used commonly by partridge, but may also be core habitat for both mammalian and avian predators. Surviving pairs were more associated with herbaceous vegetation in set aside, which is a more ephemeral habitat and may not support the types of predators found in more permanent habitats. At the local level, we see edge and set aside possibly providing more cover diversity, again suggesting predation avoidance options for those that survived.

Breeding habitat is often ignored in research on partridge ecology with more focus on winter and brood habitat. Our data suggest that habitat relative to survival during this period may impact numbers of adults going into the nesting season.

Acknowledgments

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Effects of Timber Density on Northern Bobwhite Autumn Abundance

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Mature pine (*Pinus* spp.) ecosystems maintained with frequent prescribed fire are the primary habitat of northern bobwhites (*Colinus virginianus*) in the Red Hills region of northern Florida and southern Georgia. Timber volume is thought to be negatively related to bobwhite abundance; however, this relationship has not been quantified. We related mean basal area of mature trees ($>$15cm dbh) to autumn covey call count indices at 23 locations on 6 study areas with varying timber volume, but similar bobwhite management practices, 2002 - 2004. Bobwhite abundance was inversely related to timber volume ($r = -0.61$, $P = 0.002$). Adjusted covey counts averaged 11.3 coveys below, and 6.4 coveys above, 9.2 m^2/ha (40 ft^2/ac) of basal area ($F_{1,21} = 19.4$, $P < 0.001$). Where maintaining high densities of bobwhites is a priority, we recommend pine basal areas be $<$10 m^2/ha. However, our data also suggested that bobwhites can be maintained at a bobwhite/0.4 ha at basal areas up to 14 m^2/ha assuming sound management is applied.

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Key words: abundance, autumn, basal area, *Colinus virginianus*, northern bobwhite, quail, timber

Introduction

In much of the southeastern USA, coastal plain upland-pine forests have historically been the most important habitat type for bobwhites (Stoddard 1931, Rosene 1969). Changes in pine silviculture and a reduction of prescribed fire have lowered the suitability of pine forests as habitat for bobwhites (Brennan et al. 1998, Palmer et al. 2004). A decline of early-successional habitats through conversion of open agricultural lands to closed-canopied pine forests over recent decades has further increased the significance of forested habitats for bobwhite conservation in the Southeast (Fies et al. 1992).

It has been recommended that forested landscapes be maintained with 30-50% canopy closure and less than 9.18 m^2/ha (40 ft^2/ac) of timber basal area (Rosene 1969, Brennan 1999, Burger 2001). These relatively open timber canopies, along with frequent prescribed fire, are necessary for maintaining the mix of herbaceous and woody ground cover needed to meet the life history requirements of bobwhites. It has long been thought that timber volume mediates groundstory suitability for bobwhites and is generally negatively correlated with bobwhite abundance (Stoddard 1931, Rosene 1969, Moser and Palmer 1997, Brennan 1999). While the relationship between timber density and bobwhite abundance is generally understood no effort has been made to quantify the relationship.

On southeastern bobwhite hunting plantations, bobwhite management actions and general groundstory habitat conditions in upland pine forests are relatively consistent across property ownership (Moser et al. 2002). Maintenance of groundstory vegetation suitable for sustaining high density bobwhite populations is accomplished with biennial burning in conjunction with mechanical treatments. Pine silviculture dominates the upland areas with limited midstory and/or pine regeneration (Moser et al. 2002). Unlike groundstory management, however, timber volume does vary significantly both within and across land ownerships. The similarities

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in landscape conditions among hunting plantations create an opportunity to investigate the relationship between timber density and bobwhite abundance. The trade-off between timber density and bobwhite management has important financial and biological consideration for managers (Moser and Palmer 1997, Engstrom and Palmer 2003). The objective of this study was to quantify the relationship between timber density and autumn bobwhite abundance on southeastern hunting properties managed for bobwhites.

Study Area

We selected 6 properties in the Red Hill Region of north Florida and south Georgia, including Tall Timbers Research Station (TT), Pebble Hill Plantation (PH), and 4 private plantations, DE, LL, SH, SW. Four properties (TT, PH, SW and SH) were comprised mainly of second growth loblolly and shortleaf pine uplands located on old fields (formerly agricultural fields), with <10% of overstory in longleaf pine. Two plantations were predominantly longleaf pine; 93% of timber at DE and 75% of timber at LL was longleaf pine.

The primary management objective of these plantations is to maintain high density bobwhite populations with secondary objectives including other game wildlife species, and/or timber production. Management includes use of low intensity biennial prescribed fires, roller drum chopping, and mowing to produce groundstory conditions favorable for bobwhites. In the past, timber density and composition varied among and within each plantation due to differing land uses and the interest of each owner in generating revenue from timber sales. Timber was managed using the Stoddard-Neel system, or variations of it, resulting in a mature canopy of pines over most of the uplands (Moser et al. 2002).

Methods

Research and Sampling Design

Our objective was to relate covey abundance to timber volume. We estimated bobwhite abundance using covey call point counts. The bobwhite covey call can be heard up to distances up to approximately 500 m (Wellendorf and Palmer 2005). Therefore, we chose listening points to sample covey calling and then quantified the basal area of the stand within the surrounding 500 m. We located all the possible listening points on a property such that a 500 m radius fit within the property, was composed of >75% upland pine forested habitats and contained minimal hardwood bottomland forest habitats. Among the plantations, DE, LL and PH each contained 4 survey points, SH and SW had 3 survey points and TT had 5 survey points.

Bobwhite Covey Abundance

We estimated bobwhite abundance by conducting covey call point counts during October - November, 2002 - 2004, using trained observers (Wellendorf and Palmer 2005). We counted coveys calling from 45 minutes before sunrise to sunrise (DeMaso et al. 1991, Seiler et al. 2002, Wellendorf et al. 2004). We adjusted raw counts by an estimate of the predicted calling rate to give an estimate of the number of coveys within the listening radius of the point and used this value as index of bobwhite abundance (Wellendorf et al. 2004). We conducted covey call counts at each point 1 to 3 times during each autumn, 2002 - 2004. When multiple counts were made at a point during a single autumn, the maximum covey count recorded at that point was used in analyses. To avoid temporal psuedoreplication, we then averaged corrected counts for each point across all years and used this average covey abundance in our analyses.

Estimates of Timber Density

We estimated timber basal area within the 500-m radius surrounding each survey point (78.5 ha). Sampling points were evenly spaced over the 78.5 ha area by overlaying a 2.5 ha grid in ArcView GIS 3.2. Using GPS receivers, technicians located sub-sample points at the intersection of grid lines and determined total tree basal areas using a 10-factor prism (Avery 1967). Basal area, the total estimated cross-sectional area of timber (conifer and hardwood) in a stand at breast height (approx. 1.35m), expressed in m^2/ha, was measured at each point. We generated a

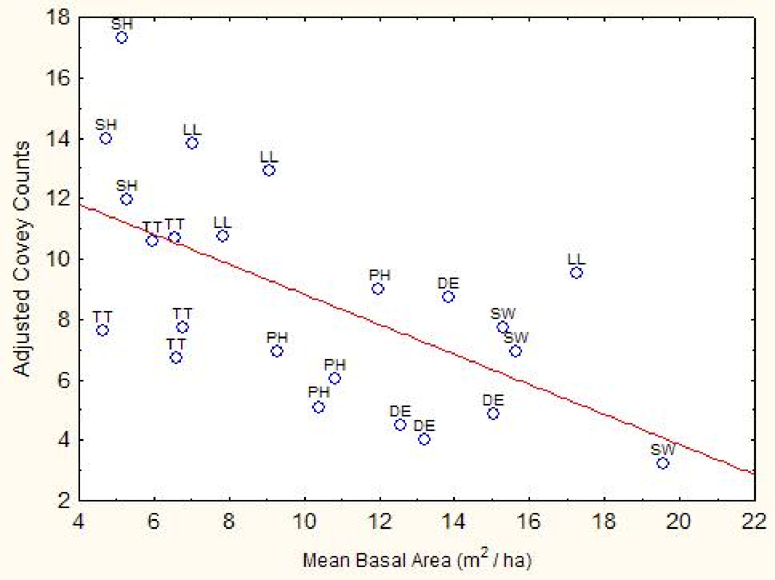

Figure 1: Relationship between mean basal area of timber (m^2/ha) within a 500-m radius surrounding northern bobwhite (*Colinus virginianus*) covey call point counts and the 3-year average of covey counts (adjusted for predicted calling rates) on 6 sites (DK, LL, PH, SH, SW, TT), Thomas and Grady counties, Georgia and Leon County, Florida, 2002 - 2004.

mean basal area from all subsample points.

We related bobwhite abundance to timber density in 3 ways. First we correlated mean basal area/point and the average number of coveys counted. However, it is unlikely that a linear model would be the logical underlying model for the affect of timber volume on bobwhite abundance. Therefore, we also considered simple curvilinear models. We hypothesized that bobwhite abundance would not be affected by timber density below some threshold but would be above this threshold. Given our limited data set and range of basal areas on our study areas, we chose to conduct a simple piecewise linear regression to determine if a breakpoint occurred in the relationship of interest. We used the Quasi-Newton estimation method to determine the best fit for our data. We then compared mean abundance of bobwhites below and above the breakpoint to provide a general effect size for timber volume on bobwhite abundance within the ranges tested in this study (Stat Soft Inc. 2003).

Results

Basal Area and Bobwhite Abundance

We surveyed bobwhite abundance at 23 locations, of which 20 points were visited each year of the study and 3 points were only visited in 2004. Our mean adjusted covey count was 8.72 (Range 3.21 - 17.29) coveys per point over the 3 year study. Basal area of timber in the 500 m surrounding each point averaged 10.2 m^2/ha (Range 4.62 - 19.56 m^2/ha or 20.1 - 85.2 ft^2/ac).

Timber densities and bobwhite abundance varied within and among the 6 study sites (Figure 1). Bobwhite abundance was inversely related to timber volume ($R^2 = 0.37$, $r = -0.61$, $P = 0.002$; Figure 1). Piecewise regression estimated a breakpoint existed in the regression line at 8.72 m^2/ha, we compared mean bobwhite abundance above and below 9.18 m^2/ha (40 ft^2/ac) to assess effect size. Bobwhite abundance was significantly higher when basal areas were below 40 ft^2/ac than above 40 ft^2/ac ($F_{1,21} = 19.4$, $P < 0.001$). Adjusted covey counts averaged 11.3 coveys below, and 6.4 coveys

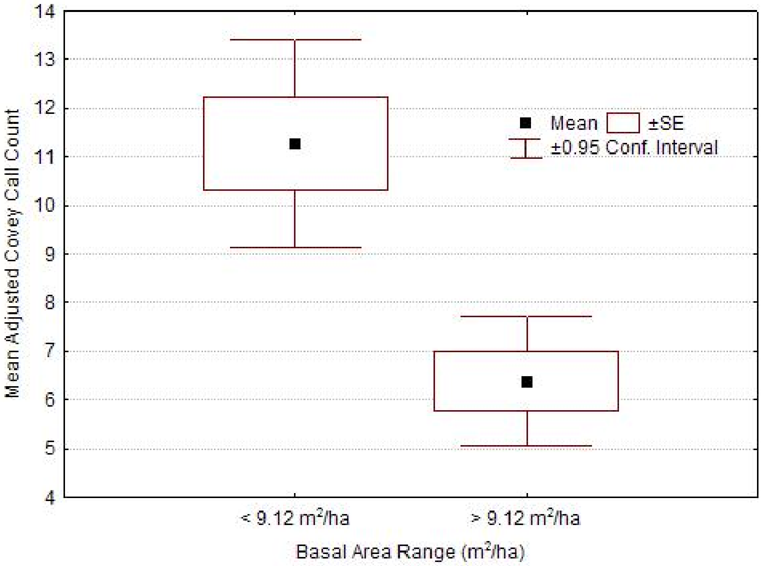

Figure 2: Mean adjusted covey call counts and associated 95% CI of northern bobwhite (*Colinus virginianus*) coveys at 11 sites with <9.18 m^2/ha (40 ft^2/ac) of timber basal area, and 12 sites >9.18 m^2/ha, Thomas and Grady counties, Georgia and Leon County, Florida, 2002 - 2004.

above, 40 ft^2/ac of basal area (Figure 2).

Discussion

Prior to this study, it had been hypothesized that bobwhite populations begin to decline at timber densities greater than 13.8 m^2/ha (Rosene 1969). Our data suggests that, where bobwhite populations are the primary objective of land management, mature pine stands should be maintained at approximately 10 m^2/ha or less.

While bobwhite abundance was greatest in timber stands below 10 m^2/ha, their abundance was still considered very good at higher timber volumes. Based on comparisons of bobwhite densities to point counts (Wellendorf and Palmer 2005), our data suggest that bobwhites can be maintained with sound management at approximately 2.5 bobwhites per ha when timber basal areas range from 10 - 14 m^2/ha (about 40 to 60 ft^2/ac). This range of timber density is also suitable for other pine forested obligate species including red-cockaded woodpeckers (*Picoides borealis*) (Engstrom and Palmer 2003). Above 14 m/ha, we expect that bobwhite populations would decline due to declining suitability of ground cover composition and structure. However, given that the areas we measured are all managed for bobwhites, few of the stands we measured were above 15 m^2/ha. Additional research needs to be conducted on stands in the 16 to 20 m^2/ha range. The greatest basal area we measured was 19.6 m^2/ha and the adjusted covey count was the lowest we measured at 3.2, which is approximately 1 bobwhite/1.6 ha. It is likely that a combination of factors influence bobwhite abundance as timber density increases, principally the suitability of the groundstory composition and structure, which likely affects predation rates, foods and foraging efficiency, and microclimate.

The replacement of frequently burned, open pine woodlands, with modern agriculture and closed pine plantations has been a primary cause for the decrease in bobwhite abundance throughout much of its distribution (Brennan 1991, Brennan et al. 1998, Palmer et al. 2004). Bobwhite managers working with upland pine forests should recognize that even with biennial prescribed fire, appropriate scale of management (10-50 ha burn units), and other management practices such as supplemental feeding and predation management, density of timber on a site has a relatively large effect on bobwhite abundance. Therefore, when planning restoration of bobwhites on a site, both reducing canopy coverage of timber and frequent prescribed fire are needed to sustain bobwhite populations.

Management Implications

We recommend on southeastern hunting plantations where maximum bobwhite populations are desired for hunting that managers maintain timber density less than 9.18 m^2/ha (40 ft^2/ac). However, this low level of timber density may not be suitable for some plantation owners due to the loss of revenue from timber sales. Within the groundstory management parameters of our study areas, huntable populations of bobwhites can still be achieved with timber densities ranging from 10 to 15 m^2/ha. While we did not measure timber density greater than 19.8 m^2/ha we would expect that bobwhite population sustainability would be greatly impacted by timber density greater than this.

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Usable Space Versus Habitat Quality in Forest Management for Bobwhites

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We determined whether habitat quality (patch richness and diversity, edge density) or the quantity of usable space (suitable permanent cover) best explained the presence or absence of calling male bobwhites (*Colinus virginianus*) on 50-ha, circular plots ($n = 80$) in the Ouachita National Forest, Arkansas, during 2000 and 2001. Information theoretic analysis of logistic regression models revealed that usable space was predictive of bobwhite occurrence, whereas habitat quality variables were not. A logistic regression model with data pooled over 2 years predicted the presence of calling males if usable space was $>$26 ha on the 50-ha plots. Creation of usable space on the study area involved removal of mid story and co-dominant shortleaf pine (*Pinus echinata*) and hardwoods (wildlife stand improvement = WSI) followed by February or March (late dormant season) prescribed burns at 3-year intervals. Application of WSI and fire results in an understory mosaic of low woody sprouts, forbs, and grasses of varying heights depending on time since the last burn. Our results were consistent with the hypothesis that predicts bobwhite presence as a function of usable space in time versus the classical principle of edge and other habitat quality indicators.

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Key words: Arkansas, *Colinus virginianus*, edge, habitat quality, northern bobwhite, pine-bluestem, prescribed burning, usable space, wildlife stand improvement

Introduction

Habitat management for bobwhites may be viewed under 2 competing hypotheses (Guthery 1997). The first hypothesis involves management for bobwhite habitat quality; it operates under the assumption that bobwhite density on a management area increases with quality attributes such as food supplies, floral diversity, cover interspersion, and edge density, among other variables. For example, management to this end would encompass, respectively, adding food plots or supplemental feeding, plantings of various types of cover (woody plantings or warm season grass plantings), and creation of additional edge within a landscape (Guthery et al. 2002). This hypothesis assumes that habitat quality exists on a continuum from low to high and further that increases in the attributes will necessarily increase bobwhite density (Guthery 1997). Leopold (1933) could be considered the originator of the habitat quality hypothesis.

The second hypothesis involves management for suitable permanent cover (usable space = structurally suitable habitat) across a landscape with less or no regard for habitat quality issues (Guthery 1997, Guthery et al. 2001). Suitable permanent cover for bobwhites is defined as persistent woody cover (<2 m in height) well mixed with herbaceous vegetation (Cram et al. 2002, Guthery et al. 2002). This hypothesis operates under the simple assumption that mean density of bobwhites on an area is proportional to the quantity of usable space on the area. This hypothesis implies that management should create usable space where none exists rather than alter perceived habitat quality in usable space that already exists. The usable space hypothesis has long been recognized (Stoddard 1931, Leopold 1933, Errington

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and Hamerstrom 1936) but perhaps not fully appreciated until more recently (Guthery 1997).

We conducted a study in the Ouachita Mountains of west-central Arkansas to determine whether selected habitat quality variables or usable space at the landscape level best predicted the presence or absence of calling male bobwhites on 50-ha, circular areas similar to Guthery et al. (2001). Landscape indices of habitat quality included edge density and patch diversity and richness. The habitat quality hypothesis predicts a positive response of bobwhites to these variables, again similar to Guthery et al. (2001). Conversely, the usable space hypothesis predicts a positive response to the quantity of suitable permanent cover and a neutral response to habitat quality variables. We examined only edge-related and patch-related aspects of habitat quality because food based aspects of the quality hypothesis and usable space hypotheses may be confounded to some degree and our interest was at a coarser scale, plus we were paralleling study design of Guthery et al. (2001) albeit in a markedly different ecosystem. As well, the creation of usable space may be associated with increased food supplies, especially in thinned and burned mature forests (Guthery 1997, Cram 2001).

Study Area

The study area was within the 60,000-ha shortleaf pine (*P. echinata*)-grassland renewal area on the Poteau Ranger District in the Ouachita National Forest, Scott County, Arkansas and associated private lands (Figure 1). The Ouachita Mountains generally run east-west with long north- and south-facing slopes. Shortleaf pine tends to dominate south-facing slopes and oaks (*Quercus* spp.) dominate north-facing slopes (Foti and Glenn 1991). Soils in the study area, which developed from sandstone and shales, are thin and drought-prone. The climate is subhumid to humid with hot summers and mild winters. The maximum annual precipitation is >150 cm and the minimum annual precipitation is <100 cm (Anonymous 1973).

The pine-grassland renewal area has been managed extensively for red-cockaded woodpeckers (*Picoides borealis*) since 1990 (Masters et al. 1996). Management consisted of wildlife stand improvement (WSI; removal of midstory and codominant pine and hardwood species). Subsequently, dormant season prescribed fire (in Feb or Mar) is applied to WSI stands on a 3-year rotation; this includes thinned and burned regeneration stands (planted or naturally regenerated to shortleaf pine). Available stands also included mature forest not subject to any treatment and unburned and unthinned regeneration stands of varying ages. Cram et al. (2002) and Masters et al. (1996, 2002) provide further details about forest management in the study area.

Our study area encompassed a matrix of unmanaged shortleaf pine-hardwood [primarily oak (*Quercus* spp.) and hickory (*Carya* spp.)] and pine-grassland restoration stands of varying age-classes managed by the U.S. Forest Service and private land. Private land management varied from dense pine-hardwood stands similar to unmanaged control stands (Cram et al. 2002) to cut-over stands and finally to open, domesticated pasture lands dominated by primarily fescue grass with occasional broomsedge bluestem (*Andropogon virginianus*). Essentially none of the private lands used prescribed fire as a management tool.

We randomly selected 80 non-overlapping plots (400-m radius; 50.3 ha) and conducted calling male counts during May-early June of 2000 and 2001 (Figure 2). We generated random coordinates of points within the bounds of our study area and buffered each point to a 400-m radius. Overlapping buffered points were discarded; we continued the process until 80-non-overlapping plots were generated. We selected 400 m as the radius of our plots because it was the same radius as Guthery et al. (2001) and we wanted to be able to draw direct comparisons. Counts involved a 3-minute listening period at plot center followed by broadcasting a female assembly call in cardinal directions at 90 dB, listening for 90 seconds, again broadcasting the call, and listening for another 90 seconds (Cram et al. 2002). The counts took place between 0600 and 1100 hours. Each site

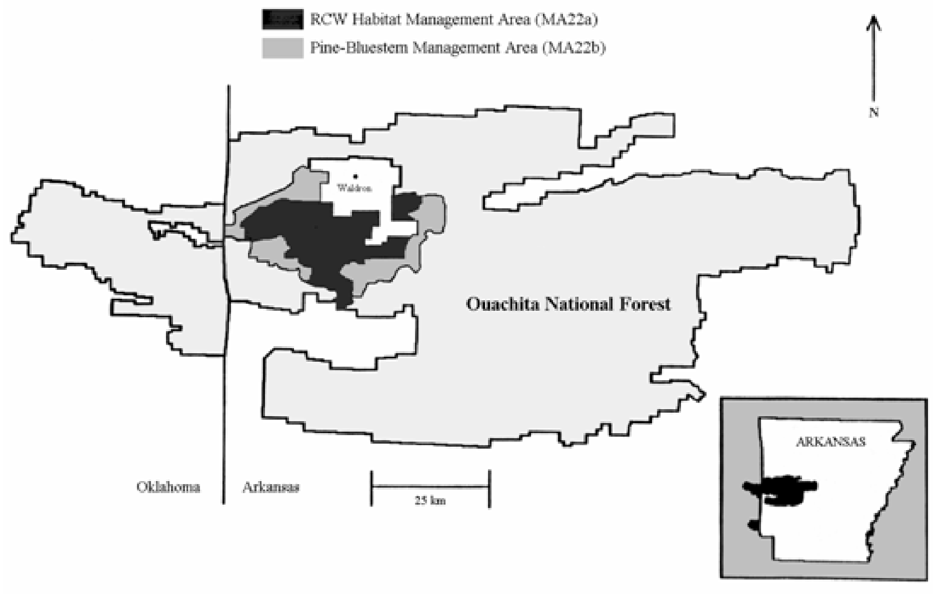

Figure 1: General study site location (MA22a and MA22b) within Ouachita National Forest near Waldron AR.

received 3 counts conducted by a different observer; one count/site was conducted during hours (0600-0800) of peak calling intensity (Hansen and Guthery 2001).

Methods

Measurement of Usable Space

Usable space (suitable habitat) availability (ha) within 400 m of a listening stop was evaluated using ArcView GIS v 3.2 (Environmental System Research Institute, Redlands, California, USA). ArcView coverages with associated stand data were obtained from the U.S. Forest Service office on the Poteau Ranger District. These maps and associated attribute tables provided areas for stand type, composition and age. Private land cover types within a listening stop were delineated from aerial photography, digitized on-screen and verified in the field for type and composition. Usable space was defined as the total area within 400 m of a listening post that consisted of stands (= habitat patch) treated with WSI only, WSI ≤3 years post fire, and regeneration stands ≤3 years post fire. The stand types defined to be usable space carry the highest density of and receive selective use from bobwhites on the study area (Cram et al. 2002, Walsh 2004). Fescue pasture was not considered usable space because early listening trials showed no calling activity across multiple years and sites and later telemetry studies (Walsh 2004) showed no use by bobwhites.

Measurement of Habitat Quality

We used FRAGSTATS (McGarigal and Marks 1994) to estimate landscape metrics for the 400 m-radius plots sampled (Table 1). Metrics such as total edge were used as indicators of habitat quality. These metrics were subjected to factor analysis (Afifi and Clark 1984) to reduce the dimensionality of the dataset.

Statistical Procedures

We used logistic regression analysis (Kleinbaum 1994) to test 7 models for predicting the presence or absence of calling male bobwhites on 50-ha areas. The global model $\{u, d, e\}$ included usable space (u; ha), Shannon diversity (d), and Shannon evenness (e). We also tested the following models: $\{u, d\}, \{u, e\}, \{d, e\}, \{u\}, \{d\}$, and $\{e\}$. Model comparisons were based on the small-sample Akaike In-

Figure 2: Listening stop locations (400 m-radius circles, 50.3 ha) on the Poteau Ranger District, Ouachita National Forest and private lands near Waldron AR, May through early June 2000 and 2001.

formation Criterion (AIC$_c$; Burnham and Anderson 2002). The data from 2000 were used to generate models and the data from 2001 were used to test the repeatability of the best model from the 2000 data. We also report a model for data pooled over the 2 study years.

Results

Of the 80 listening stops, 26 encompassed at least some private land or were entirely on private land. Our listening stops encompassed a total of 12 different habitat types. Metrics associated with characteristics of discrete habitat patches (quality variables) within a given listening stop (e.g., number of patches, patch density, and patch richness) had a generally wide range about the mean (Table 1). Two factors explained 89% of the variation in the landscape metrics (Table 2). Variables with high loadings (absolute value) on the first factor included patch diversity, patch richness, and total edge, among others. Evenness indices and contagion loaded high on the second factor. These results indicated the landscape metrics evaluated collapsed to essentially 2 variables because of intercorrelation among variables (variables with high loadings on the same factor are correlated). Therefore, we used Shannon diversity (highest loading on first factor; 0.95) and Shannon evenness (second highest absolute loading on second factor; -0.86) in logistic modeling (variables with high loadings on different factors are not correlated).

During both 2000 and 2001, the average number of calling males/stop was 0.2 ± 0.04 SE ($n = 80$ each year). In 2000 calling males were recorded at 30 of 80 stops, whereas in 2001 they were recorded at 27 of 80 stops. Over the 2 years, 41 listening stops had calling males in $\geq$1 year.

Model selection revealed that any model that contained usable space was plausible ($\Delta\text{AIC}_c < 1.4$), whereas any model that did not contain usable space was not plausible ($\Delta\text{AIC}_c > 9.4$; Table 3). The Akaike best model contained usable space as the sole variable. Given presence of bobwhites, this model correctly predicted 45% of cases; given absence, it correctly predicted 67% of cases.

The logistic regression model (u only) generated with 2000 data correctly predicted 61% of cases for

Table 1: Mean values for landscape metrics of 50 ha, circular plots ($n = 80$), Ouachita National Forest, Scott County, Arkansas, 2000 and 2001.

Landscape Metric	Mean	Min	Max	Range
Number of patches	10.56	2.00	18.00	16.00
Patch density (no./100 ha)	21.12	4.00	36.00	32.00
Landscape patch index (%)	35.84	15.67	86.89	71.22
Total edge (m)	4879.38	1115.00	8720.00	7605.00
Edge density (m/ha)	97.58	22.30	174.40	152.10
Landscape shape index	2.86	1.53	4.21	2.69
Contagion	57.56	47.50	81.72	34.22
Cohesion	98.82	98.06	99.71	1.65
Patch richness	8.68	2.00	18.00	16.00
Patch richness density (no./100 ha)	17.35	4.00	36.00	32.00
Shannon diversity	1.64	0.59	2.21	1.62
Simpson diversity	0.75	0.24	0.87	0.63
Modified Simpson diversity	1.45	0.28	2.07	1.79
Shannon evenness	0.79	0.33	1.00	0.67
Simpson evenness	0.86	0.29	1.00	0.71
Modified Simpson evenness	0.70	0.15	1.00	0.85

the 2001 data, despite the fact that some stops with calling males in 2000 had no calling males in 2001, and some stops with no calling males in 2000 had calling males in 2001. Given presence of calling males, the model correctly predicted 38% of cases in 2001. Given absence, it correctly predicted 72% of cases.

The logistic regression model for data pooled over years was

$$y = \frac{1}{1+exp(1.04-0.04x)}$$

where y is the classification score and x is usable space (ha). The 95% CI on the intercept (1.04) and coefficient (0.04) did not contain zero. This model predicts the presence of calling male bobwhites if the quantity of usable space exceeds about 26 ha ($y > 0.5$). Bobwhite occurrence plots ($n = 41$) were associated with an average of 34 ha of usable space (95% CI = 29.7-36.3 ha), whereas non-occurrence plots ($n = 39$) were associated with 21 ha (95% CI = 16.7-25.3 ha). Given presence, the model predicted correctly in 57% of cases, whereas given absence, it predicted correctly in 55 of cases.

Discussion

As with all field studies, our results have contexts for interpretation and extrapolation. First, the results pertain specifically to calling male bobwhites because we collected no data on other sex-age classes. However, Cram et al. (2002) observed that pine-grassland restoration (WSI, WSI plus burning) provided suitable habitat structure for bobwhites during spring, summer, and fall but may not be adequate during winter. Walsh (2004) observed bobwhite selection for pine-grassland restoration treatments and thinned and burned regeneration stands in both the breeding and covey season. Our listening-stop data also confirmed that domesticated fescue (*Festuca* sp.) pasture was not usable space. Thus, our findings generally were consistent with previous findings on the study area.

Second, our experiment took place in a low-density population. During both 2000 and 2001, the average number of calling males/stop was 0.2 ± 0.04

Table 2: Factor loadings for landscape metrics associated with 50-ha, circular plots ($n = 80$), Ouachita National Forest, Scott County, Arkansas, 2000 and 2001.

	Factor	
Landscape metric	I	II
Number of patches	0.83	0.48
Patch density (no./100 ha)	0.83	0.48
Landscape patch index (%)	-0.85	0.37
Total edge (m)	0.88	0.22
Edge density (m/ha)	0.88	0.22
Landscape shape index	0.88	0.22
Contagion	-0.52	0.82
Cohesion	-0.88	-0.31
Patch richness	0.82	0.43
Patch richness density (no./100 ha)	0.82	0.43
Shannon diversity	0.95	0.01
Simpson diversity	0.91	-0.23
Modified Simpson diversity	0.94	-0.17
Shannon evenness	0.47	-0.86
Simpson evenness	0.65	-0.86
Modified Simpson evenness	0.43	-0.88

SE ($n = 80$ each year). On farms and ranches in central and western Oklahoma, Guthery et al. (2001) observed an average of 4.2 ± 0.43 calling males/stop with a range of 0.1-15.9 males/stop during 1998 and 1999. The apparent low densities we observed could be related to timing of the counts (May-early June). Hansen and Guthery (2001) observed peak calling of bobwhite males in mid June or later in central Oklahoma. Terhune (2004) reported peak calling in June and July in south Georgia and north Florida.

Third, the predictive models developed, though statistically well supported (Table 3), were of relatively low accuracy. For example, the model for data pooled over years correctly predicted 56% of cases overall. We conjecture that the relatively low accuracy occurred for at least 3 reasons. First, population density of calling males might have been too low to fully occupy usable space (more usable space than males available). This situation would have been associated with the absence of males on suitable 50-ha circles, thus resulting in modeling score of 0 (absence) in suitable cover. This is plausible because changes in bobwhite density lag somewhat behind creation of suitable habitat because a response is time dependent (Guthery 1997). This was evidenced by comparison of bobwhite abundance just after initiation of the pine-bluestem renewal project by Wilson et al. (1995) and a study a decade later by Masters et al. (2002) within the pine-bluestem renewal area. The amount of treated area on the forest was 1,057 ha in 1993 and 4,970 in 2001; an increase of almost 5 times the usable space. Second, in our study area, bobwhite density in a stand is related to the amount of suitable habitat surrounding the stand (Cram et al. 2002). Our limit of analysis (400 m-radius) did not fully capture this potentiality, which would imply the possible occurrence of calling males in 50 ha plots with low quantities of usable space if larger quantities of usable space were nearby. Third, all treatment types on the study

Table 3: Evaluation of logistic regression models for predicting the presence or absence of calling male bobwhites on 50-ha, circular plots (n = 80), Ouachita National Forest, Scott County, Arkansas, 2000 and 2001.

Model[a]	K	AIC_c	ΔAIC_c	Proportion correct
u,d,e	4	100.4	0.50	0.61
u,d	3	100.7	0.80	0.59
u,e	3	101.2	1.34	0.59
d,e	3	110.7	10.81	0.54
u	2	99.9	0.00	0.59
d	2	109.3	9.44	0.54
e	2	109.8	9.98	0.53

[a] u = usable space (ha), d = Shannon diversity, e = Shannon evenness

area, including untreated forest, contain at least limited amounts of usable space (Walsh 2004). This would imply usable space as we defined it (WSI, WSI plus burning, and burned and thinned regeneration stands) should be considered an index of the total quantity present. This situation could reduce the accuracy of prediction models.

Given limitations on predictive accuracy, however, the models and other analyses were useful in revealing patterns in the dataset. Factor analysis revealed that several variables presumed indicative of habitat quality measured essentially the same property of experimental plots. The number of patches, total edge, patch richness, and diversity indices all loaded high on Factor I (Table 2). This means the variables were intercorrelated and therefore, at least on our study area, knowing any 1 of them provided information on all the others. Note that landscape metrics expressed as density (e.g., patch density) were scaled versions of the raw homologue (e.g., number of patches) because all areas were 50 ha in our study and thus had factor loadings identical to the homologues.

The pattern revealed by logistic regression models was that male bobwhites responded to the availability of usable space but not to patch diversity and related variables (e.g., total edge; Table 3). This result supports the usable space hypothesis over the habitat quality hypothesis. Likewise, Guthery et al. (2001) found stronger support for the usable space hypothesis than for the habitat quality hypothesis on farms and ranches in central and western Oklahoma. These authors observed a negative effect of Shannon diversity, patch richness, and density of woody edge on the abundance of calling male bobwhites.

However, the habitat quality and usable space hypotheses are confounded to some degree. For example, the existence of edge between woody cover and prairie implies usable space (Guthery and Bingham 1992) for bobwhites. Within usable space, however, bobwhite density may be independent of the quantity of edge (Guthery 1999, Guthery et al. 2001). Also, the creation of usable space may be associated with increased food supplies, especially in thinned and burned mature forests (Guthery 1997, Cram 2001).

Management Implications

Our results on the presence or absence of calling male bobwhites supported the usable space hypothesis over aspects of the habitat quality hypothesis for management of bobwhite habitat in the Ouachita National Forest, Arkansas; specifically as it relates to patch characteristics of edge. Management that creates usable space in this setting (mostly contiguous dense forest) implies thinning of young and mature stands of timber to a basal area of at least 13.8 to 18.4 m^2/ha, followed by prescribed burning of thinned stands on a $\leq$3-year rotation (Cram et al. 2002). This reduction in stand basal area results in an increase in understory vegetation composed of a mosaic of low woody sprouts, forbs and grasses of varying heights depending on time since the last burn (Cram et al. 2002, Masters et al. 1996). The increase in understory vegetation also apparently increased food supplies (plant density and thus seed production, plus insect abundance; Cram 2001). Woody vegetation density <2m was an important predictor of bobwhite abundance (Cram et al. 2002). Longer burning rotations than 3-year intervals do not appear acceptable based on other research in this area as rapid understory woody development soon creates unsuitable habitat structure (Cram et al. 2002, Walsh 2004). Further our results provide additional evidence that fescue pasture is not usable space.

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Influence of Habitat, Fire, and Weather on Bobwhite Abundance at Avon Park Air Force Range, Florida

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From 1993 to 2005, we conducted 13 years of early breeding season call counts at Avon Park Air Force Range, in south-central Florida. Historically, this area has been an open, frequently burned landscape with a rainy summer season and a dry spring. We used call count data (*n* = 67 stations/year) and 400-m buffer areas around each point to investigate the effects of habitat preference, fire impacts, and weather variations on bobwhite abundance. Using logistic regression, we determined that bobwhite in south-central Florida prefer open canopy, dry/mesic habitats in prairie and flatwoods over pine plantations and wet areas. Investigated by ANOVA, bobwhite abundance was highest when at least 40% of a buffer area was burned. Using only burns which occurred on at least 40% of a buffer area, higher bobwhite abundance occurred in buffers which received a dormant or growing season burn within the previous year. Two years post-burn, bobwhite abundance significantly decreased for both burn seasons. Burning in less optimal habitats (e.g., those dominated by dense canopy) did not affect bobwhite populations. Correlation analysis revealed significant negative correlations between bobwhite abundance and April PMDI, May PMDI, and November rain. Bobwhite abundance had significant positive correlations with October PMDI and July rain days. By knowing how local habitat, fire, and weather can impact early breeding season bobwhite abundance, managers may help their population flourish.

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Introduction

In the Southeast, the northern bobwhite (*Colinus virginianus*) populations have been declining for decades (Brennan 1991, Capel et al. 1995, Peterson et al. 2002, Brennan and Kuvlesky 2005, Stewart 2005). Changes in land use, reduction in the use of fire, and reduction of usable space and habitat have been attributed to its decline (Roseberry et al. 1979, Brennan 1991, Guthery 1997, Brennan et al. 1998, Brennan and Kuvlesky 2005, Stewart 2005). A better understanding of which habitats and management actions, such as fire, can promote productive bobwhite populations is important, as well as knowing how local weather may impact a population.

Throughout their range, bobwhites depend on early successional habitats in high productivity sites to late successional habitats in sites of low productivity for successful reproduction and survival (Spears et al. 1993, Guthery 1997). Generally, where there is an open canopy (if any), a sparse woody mid-story, and a diverse groundcover layer, bobwhites may be abundant (Landers and Mueller 1986, DeVos and Mueller 1993, Cram et al. 2002, Stewart 2005). As canopy cover increases in woodlands, bobwhite abundance declines (Lee and Brennan 1994, Cram et al. 2002). Without periodic disturbance, such as fire, microhabitat characteristics become unsuitable for bobwhites during the breeding season (Landers and Mueller 1986, Stewart 2005).

In the Southeast, fire has occurred for many centuries, historically during the early lightning season (Komarek 1964, Christensen 1981, Abrahamson and Hartnett 1990); thus, bobwhites have evolved with this natural disturbance. Bobwhites may benefit from fire by it reducing litter and woody plant invasion and increasing insect abundance and plant forage (Murray and Frye 1964, Hurst 1972, Moore 1972, Landers 1981, Landers and Mueller 1986, Lewis and

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Harshbarger 1986, Brennan et al. 1998, Hermann et al. 1998). On the other hand, fire may reduce nest sites and escape cover and consume nests and chicks (Dimmick 1972, Landers 1981, Robbins and Myers 1992, Lyon et al. 2000*a*,*b*). When fire is excluded from these fire-adapted plant communities, bobwhite populations may decrease over time (Engstrom et al. 1984).

Season of fire can have differing effects (Landers 1981, Robbins and Myers 1992, Hermann et al. 1998). Fires conducted during the winter may reduce woody plant cover for only a few months but may actually increase number of stems by sprouting (Waldrop et al. 1987, Robbins and Myers 1992). Spring and early summer fires may more effectively reduce woody plant cover and abundance and increase bare ground, especially with annual burns (Waldrop et al. 1987, Robbins and Myers 1992, Brennan et al. 1998, Hermann et al. 1998).

Weather has been shown to significantly influence bobwhite abundance by directly or indirectly affecting survival and productivity (Speake and Haugen 1960, Jackson 1962, Stanford 1972, Bridges et al. 2001, Perez et al. 2002, Hernandez et al. 2005). Excessive rains may cause flooding which can drown adults and chicks and kill or reduce vegetation for forage or cover (Frye 1948, Rosene 1969, Landers and Mueller 1986). Droughts can reduce productivity by reducing hatching rates, nesting rates, percentage of hens nesting, cover, and forage (Speake and Haugen 1960, Jackson 1962, Stanford 1972, Landers and Mueller 1986, Hernandez et al. 2005). In the Southeast, wet summers can yield greater bobwhite reproductive output, whereas dry (especially hot) summers can yield lower reproductive output (Murray 1959, Reid and Goodrum 1960, Speake and Haugen 1960). On poorly drained, flat landscapes, very wet summers or hurricanes have had a negative impact on bobwhite populations by drowning nests, chicks, and adults (Frye 1948).

Few studies have investigated northern bobwhites in south-central Florida for habitat, fire, and weather influences on bobwhite breeding abundance. In an effort to better understand which habitats may be optimal and what effects fire and weather may have on bobwhite breeding abundance, data from thirteen years of breeding season call-counts were used to determine any influence they had on bobwhite breeding abundance.

Study Area

The study area was located on Avon Park Air Force Range (APAFR, 42,430 ha, 27° 37′ N, 81° 16′ W), Highlands and Polk counties, which is in south-central Florida, USA. APAFR is managed for multiple uses including military activities, timber resources, cattle grazing, natural areas, endangered species, and recreation uses, including hunting. The main plant communities in the study area include pine flatwoods, wet and dry prairies, and pine plantations, with some swamps, marshes, hammocks, scrub, and improved pastures. An endemic flatwoods community occurs on the installation called cutthroat grass (*Panicum abscissum*) flatwoods, which is dominated by the dense, coarse cutthroat grass (Bridges and Orzell 1999). The climate is characterized as humid subtropical, with mild, dry winters and hot, wet summers (Chen and Gerber 1990). The temperatures range from a monthly mean January temperature of 16°C to a monthly mean August temperature of 28°C. Most of the mean yearly rainfall of 127 cm falls from June to September (72 cm, 57%).

Burning has never ceased on the installation, but the season in which it is has burned has shifted from natural regimes (S. L. Orzell, APAFR, unpublished data). Ordnance fires can occur on the ranges at any time of the year. In south-central Florida, the natural season for fire occurrence was in the late spring to early summer, during the time period when there is usually an ongoing drought and lightning storms start occurring (Komarek 1964, Snyder 1991, Beckage et al. 2003, Slocum et al. 2003, W. J. Platt, Louisiana State University, unpublished data). The managers have been shifting some prescribed burning back to this time period over the last decade. During the study period most of the burns were during January (30%) and February (21%), followed by May (12%) and March (11%).

Methods

Call Counts

Since 1993, annual male whistle counts have been conducted for 7-9 weeks from April to June on seven permanent routes, which had 11-12 listening-stations each spaced ca. 0.8 km apart, for a total of 83 stations. The routes were distributed in a fairly even manner across the installation with no regard for habitat. The counts started around sunrise and continued for about 2 hours past sunrise until a given route was completed. The number of "bobwhite" calls and number of bobwhite calling for 8 minutes per station were recorded. Each route was repeated weekly during the time period with the order of stations reversed each time.

Since peak call rates coincide with peak breeding initiation activity and thus detection rates increase (Hansen and Guthery 2001), the four highest replications for a given route in bobwhite number were used for determining average bobwhite per station per year. In most years, the replications used were mostly in May with some weeks in early June and late April.

Listening Station Attributes

All stations were fixed with a global positioning system and a layer in ARCGIS created. In ARCGIS, we created 400-m buffers around each station, in order to obtain habitat and burn histories. Four hundred meters is within the range of the assumed audible distance of the "bobwhite" call for the human ear (Hansen and Guthery 2001). We discarded stations that overlapped more than 5% within the 400-m buffer from the analysis due to the possibility of non-independence of station data. We discarded 4 other stations because they bordered the property, so their management history was unknown. This left a total of 67 stations for analysis.

Habitat

Within each buffer, we determined the proportion of each habitat type based on a plant community GIS coverage created by Edwin Bridges (Table 1). Some habitats were further broken down by wet and dry/mesic classes and closed (moderate to dense closure) and open (none to sparse closure) canopy classes (Table 1). Based upon the dominant habitats (≥20% of the buffer area) and canopy closure of the optimal habitat (as determined from the analysis), we classified each station into five habitat groups. The five habitat groups were *OpOpt*, dominated by open canopy dry/mesic flatwoods, prairie, and/or cutthroat only (*n* = 17); *OptPl*, dominated by plantations and open canopy dry/mesic flatwoods, prairie, and/or cutthroat (wet habitat may be present, *n* = 18); *OptWet*, dominated by wet habitats and open canopy dry/mesic flatwoods, prairie, and/or cutthroat (*n* = 11); *ClOpt*, dominated by closed canopy dry/mesic flatwoods, prairie, and/or cutthroat only (*n* = 9); *Other*, dominated by habitats other than dry/mesic flatwoods, prairie, and/or cutthroat (*n* = 12). Mean bobwhite per habitat group per year were calculated for habitat group analysis. All means are given with ±1 SE.

Fire

Within each buffer, we determined the season of burn, proportion of area burned, and length of time between burns for the sample period. We categorized burn types into *dormant season burns* (DSB, from October of previous year to March of current year) and *growing season burns* (GSB, from previous year's April through September). Because of when the call counts were conducted, GSB effects were not assessed for almost a year after they happened because the burn may have occurred during or after the call count survey period. During the time of the study, most GSB occurred in May (39%), followed by June (26%) and April (21%). Therefore, we analyzed DSB and GSB separately.

We categorized stations by the proportion of area burned each year into the following categories: 0%, 10% (1-19%), 30% (20-39%), 50% (40-59%), and 80% (60-100%). To avoid confounding effects between seasons, 0% burns only represent incidents where neither a DSB nor GSB occurred. Due to sample sizes, the 10% and 30% categories and 50% and 80%

Table 1: Individual Wald P-values, coefficient sign, and percent of each habitat represented in the buffer areas with low abundance ($<$ 2 bobwhite on average per station,n = 45) and high abundance ($\geq$ 2 bobwhite on average per station, n = 22) for variables used to determine optimal habitat logistic regression models at Avon Park Air Force Range, Florida, 1993-2005.

			% Habitat Abundance	
Habitat	Wald P-value	Coeff. sign	Low	High
Cutthroat	0.095	+	2.2	8.3
Dry/Mesic cutthroat	0.084	+	0.5	7.9
Open canopy dry/mesic cutthroat	0.109	+	0.5	6.7
Closed canopy dry/mesic cutthroat	0.980	+	0.0	1.1
Wet cutthroat	0.809	-	2.0	1.6
Flatwoods	0.262	-	45.6	37.5
Open canopy flatwoods	0.660	+	29.1	31.8
Closed canopy flatwoods	0.033	-	16.5	5.7
Dry/Mesic flatwoods	0.977	-	33.9	33.7
Open canopy dry/mesic flatwoods	0.133	+	21.0	29.7
Closed canopy dry/mesic flatwoods	0.058	-	12.9	4.0
Wet flatwoods	0.016	-	11.7	3.8
Open canopy wet flatwoods	0.052	-	8.1	3.2
Closed canopy wet flatwoods	0.065	-	3.6	0.5
Hammocks	0.370	-	1.4	0.2
Marsh and swamps	0.423	-	12.9	10.6
Improved pasture	0.415	-	4.8	1.3
Pine plantations	0.002	-	25.3	5.5
Open canopy pine plantations	0.547	+	0.6	1.2
Prairies	0.002	+	5.3	29.1
Open canopy prairies	0.004	+	4.6	27.5
Closed canopy prairies	0.499	+	0.8	1.5
Dry/Mesic prairies	0.004	+	2.1	25.9
Open canopy dry/mesic prairies	0.010	+	1.4	24.8
Closed canopy dry/mesic prairies	0.583	+	0.7	1.3
Wet prairies	0.990	+	3.2	3.3
Open canopy wet prairies	0.890	-	3.2	3.0
Closed canopy wet prairies	0.355	+	0.1	0.3
Roads and disturbed areas	0.016	+	4.5	11.4
Scrub	0.514	+	2.0	3.2
Wet areas	0.011	-	34.1	18.7
Open canopy wet areas	0.175	-	21.2	15.9
Closed canopy wet areas	0.006	-	12.9	2.8
Open canopy areas	0.000	+	52.9	88.1

categories were pooled when fire influences for each habitat group were examined.

Based on the results of this analysis, we investigated time since burn when at least 40% of the buffer area was burned. Time since burn was broken down into two to four classes, depending on the sample size of each class. For DSB, classes were < *0.5 year, 1.5 years, 2.5 years, and 3.5 plus years* since the burn event. For GSB, classes were < *1 year, 2 years, and 3 plus years* since the burn event. For burns in each habitat group, we reduced DSB classes by one, while we reduced growing season classes by one only for habitat group *OptWet*. We pooled habitat groups *ClOpt* and *Other* for this analysis, creating the habitat group *NoOpt*. Mean bobwhite per station per burn season and class per year were determined and repeated for each habitat group. All means are given with ±1 SE.

Weather

We used raw precipitation (rain), number of days with >2.5 mm rain (rain days), and Palmer Modified Drought Index (PMDI) to determine weather effects on breeding season bobwhite abundance overall and after DSB and GSB. Palmer's Drought Severity Index takes into consideration precipitation, temperature, evapotranspiration index, runoff, soil recharge, and average regional weather conditions to measure departures from normal regional moisture supply, with the PMDI representing better real-time conditions and transitional periods (Heddinghaus and Sabol 1991). We determined raw precipitation and rain days from on-site weather stations. We obtained PMDI from NOAA's National Climatic Data Center for region 4 (south-central) in Florida. For overall abundance and abundance after a DSB, we calculated monthly, seasonal sums, and total yearly sums from previous year's April to current year's March. For abundance after a GSB, we used the previous year's October (before the burn) to the year's September after a GSB.

Data Analyses

We determined optimal habitat by forward stepwise logistic regression using Systat 7.0 (SPSS Inc. 1997). Used as the dependent variable, the stations were classified into two classes based on the mean number of bobwhite per station over all years: with less than two present (low abundance, n = 45) or with ≥ two present (high abundance, n = 22). We used the proportions of habitats within the buffers as independent variables. Broad habitat classes were used first, and then subclasses that are more distinct were used including different canopy closures within subclasses (Table 1). An alpha level of 0.15 was used for inclusion in model and 0.20 for removal from each model (Hosmer and Lemeshow 1989, 108). These values were used to avoid leaving out significant habitats as can happen in stepwise regression, especially if significance levels are too stringent. We determined each model's goodness of fit by using Hosmer-Lemeshow statistic (C, higher values mean higher significance of model) and percent correct. We also assessed each habitat class individually with the Wald statistic *p*-value and coefficient sign.

We analyzed habitat group and burn analysis by ANOVA with year as a main effect to account for yearly fluctuations. Bobwhite abundance was square-root transformed to meet assumptions of ANOVA. If there was a significant effect ($P < 0.05$), multiple range tests were conducted using least significant differences.

Weather variables were analyzed with correlation analysis using Pearson product-moment correlation coefficient. Some variables were log-transformed to meet the assumptions of normality. If transformations were not effective, spearman rank correlation was used (wet season rain days, dry season rain, and September rain days). Significance level was set at 0.10 for this analysis.

Results

During the 13 years of call counts, a total of 54242 calls and 5733 bobwhite were recorded ($\bar{x}$ = 14.7 ± 0.37 and $\bar{x}$ = 1.6 ± 0.03 per station, respectively). The highest abundance occurred in 2002 ($\bar{x}$ = 2.2 ± 0.25), followed by 1996 ($\bar{x}$ = 2.0 ± 0.17) and 2005 ($\bar{x}$ = 1.8 ± 0.16). The lowest abundance occurred in 1998 ($\bar{x}$ = 1.2 ± 0.12) and 1999 ($\bar{x}$ = 1.2 ± 0.11), followed by 2004 ($\bar{x}$ = 1.4 ± 0.14). The mean peak of calling rate occurred on Julian day 133 ± 0.5 (13 May), with the 2002 having the earliest mean calling peak ($\bar{x}$ = 122 ± 2.1, 2 May) and 2003 having the latest mean calling peak ($\bar{x}$ = 145 ± 1.7, 25 May).

Table 2: Optimal habitat model parameters, coefficients, odds ratios, Hosmer-Lemeshow statistic (C) and percent total correct as determined by logistic regression at Avon Park Air Force Range, Florida.

Model	Parameter[a]	Coeff. (SE)	Odds Ratio (95% CI)	C	Total Correct
1	Intercept	-4.00 (1.184)		0.221	73.1%
	CT	0.07 (0.025)	1.07 (1.123 - 1.019)		
	FL	0.04 (0.017)	1.04 (1.072 - 1.003)		
	PR	0.10 (0.027)	1.10 (1.160 - 1.043)		
2	Intercept	-2.81 (1.242)		0.339	84.2%
	OP PR	0.15 (0.052)	1.12 (1.292 - 1.054)		
	CCWA	-0.36 (0.172)	0.70 (0.977 - 0.500)		
	CT	0.14 (0.055)	1.15 (1.286 - 1.036)		
	OP FL	0.04 (0.022)	1.04 (1.089 - 1.001)		
3	Intercept	-5.52 (1.550)		0.370	81.8%
	DM CT	0.24 (0.102)	1.28 (1.559 - 1.044)		
	DM FL	0.07 (0.024)	1.07 (1.121 - 1.021)		
	DM PR	0.15 (0.043)	1.16 (1.264 - 1.068)		
4	Intercept	-5.73 (1.453)		0.563	85.7%
	OP DM CT	0.17 (0.090)	1.19 (1.419 - 1.001)		
	OP DM PR	0.22 (0.069)	1.25 (1.431 - 1.093)		
	OP DM FL	0.10 (0.028)	1.10 (1.166 - 1.044)		

[a] CT= Cutthroat, FL=Flatwoods, PR=Prairie, OP=Open Canopy, CCWA=Closed Canopy Wet Areas, DM= Dry-Mesic

Habitat

All of the optimal habitat models included flatwoods, cutthroat, and prairies (Table 2). Closed canopy flatwoods, closed canopy dry/mesic flatwoods, closed and open canopy wet flatwoods, wet areas, closed canopy wet areas, and pine plantations were not preferred habitat (Table 1). The most optimal habitat included open canopy dry/mesic cutthroat, open canopy dry/mesic prairies, and open canopy dry/mesic flatwoods. All models were significant, but model 4 yielded the best total correct (86%) based on the actual values versus expected values and a Hosmer-Lemeshow statistic of 0.563. Over 78% of the high stations and 89% of the low stations were correctly predicted by the model.

More bobwhite abundance significantly occurred in the *OpOpt* group ($\bar{x}$ = 2.4 ± 0.14) than the other habitat groups during the breeding season ($F_{4,48}$ = 78.38, $P < 0.001$). Wet areas (*OptWet*) were preferred to pine plantations (*OptPl*) as a codominant habitat type ($\bar{x}$ = 1.7 ± 0.11 and $\bar{x}$ = 1.4 ± 0.08, respectively). The least amount of bobwhite occurred at stations which had less than 20% of optimal habitat present with either closed canopy dry/mesic habitats or other habitats ($\bar{x}$ = 1.0 ± 0.07 and $\bar{x}$ = 1.0 ± 0.07, respectively).

Fire

Proportion Burned - More bobwhite abundance occurred when at least 40% of the buffer area was DSB or GSB ($F_{4,46} = 7.56$, $P < 0.001$; $F_{4,43} = 6.17$, $P < 0.001$, respectively; Table 3). For DSB, almost twice as many bobwhite were found at stations which had 60%+ of their buffers burned than stations which received no burns. The GSB impact was greatest when 40 to 59% of the buffer area burned.

Table 3: Mean bobwhite per station with dormant and growing season burns by percent buffer area burned and time since burn at Avon Park Air Force Range, Florida, 1993-2005.

	Dormant season burn			Growing season burn		
	#	SE	n[a]	#	SE	*n*
% buffer area burned, midpoints						
0	1.1C[b]	0.10	349	1.1BC	0.10	349
10	1.2C	0.11	86	0.9C	0.12	92
30	1.5BC	0.20	88	0.9C	0.14	83
50	1.8AB	0.19	104	1.7A	0.17	55
80	2.2A	0.32	71	1.4AB	0.22	30
P	< 0.001			< 0.001		
Years since burn						
< 1[c]	1.9A	0.22	175	1.7A	0.16	85
1-2	1.6A	0.13	161	1.5A	0.30	54
2-3	1.1B	0.15	101	0.9B	0.19	103
3+	0.7B	0.14	175			
P	< 0.001			0.026		

[a] number of occurrences in each category

[b] column means followed by the same letter or without letters were not different, $P > 0.05$ (LSD), for % buffer area burned and years since burn, respectively

[c] dormant season burns are closer to midpoint in years, while growing season burns are closer to upper value. Interval 2-3 for growing season burns represents all years beyond 2 years since burn. Burns occurred when at least 40% of buffer area was burned

When investigating burn effects within each habitat group, only *OpOpt* and *OptPl* had significantly higher bobwhite abundance when greater than 1% and 40% of a buffer area had a DSB, respectively ($F_{2,24}$ = 13.26, $P < 0.001$; $F_{2,23}$ = 6.22, $P = 0.007$, respectively; Table 4). *OptWet* had almost twice as many bobwhite at stations which were burned greater than 40% than at stations which were not burned, though it was not significant ($F_{2,17}$ = 3.26, $P = 0.063$). Only the *OptWet* habitat group had significantly more bobwhite abundance when at least 40% of the buffer area had a GSB ($F_{2,16}$ = 4.19, $P = 0.034$; Table 5).

Time Since Burn - Considering only time since burn for burns which had at least 40% of their buffer burned, bobwhite abundance increased within one year of a DSB or GSB by as much as 171% and 89%, respectively, then significantly decreased after 2 or more years post burn ($F_{3,36}$ = 11.10, $P < 0.001$; $F_{2,23}$ = 4.30, $P = 0.026$, respectively; Table 3). Higher bobwhite abundance followed DSB than GSB, though not a significant amount ($P = 0.337$).

After a DSB occurred, only *OpOpt* and *OptWet* habitat groups significantly increased in bobwhite abundance (65% and 250%, respectively), which significantly decreased after 1.5 years post burn ($F_{2,21}$ = 4.69, $P = 0.020$; $F_{2,18}$ = 9.69, $P = 0.001$, respectively; Table 4). After a GSB occurred, only *OpOpt* habitat group significantly increased in bobwhite abundance (156%), which significantly decreased after 2 years post burn ($F_{2,16}$ = 6.36, $P = 0.009$; Table 5).

Weather

Of the seasonal and yearly weather variables, only wet season rain days was significantly corre-

Table 4: Mean bobwhite per station after dormant season burns by percent buffer area burned and time since burn for each habitat group at Avon Park Air Force Range, Florida, 1993-2005.

	Habitat group[a]											
	OpOpt			*OptPl*			*OptWet*			*NoOpt*		
	#	SE	n[b]	#	SE	n	#	SE	n	#	SE	n
% buffer area burned, midpoints												
0	1.9B[b]	0.12	91	1.0B	0.15	81	1.1	0.19	58	0.8	0.11	119
20	2.6A	0.22	48	0.8B	0.16	58	1.5	0.35	16	0.9	0.14	52
70	2.8A	0.26	59	1.7A	0.31	38	2.1	0.35	37	1.1	0.27	41
P	< 0.001			0.007			0.063			0.352		
Years since burn[c]												
< 0.5	2.8A	0.26	59	1.7	0.31	38	2.1A	0.35	37	1.1	0.27	41
1.5	2.4A	0.28	42	1.4	0.21	44	1.7A	0.32	29	1.1	0.15	46
2.5+	1.7B	0.17	61	0.9	0.13	66	0.6B	0.19	46	0.6	0.09	103
P	0.02			0.054			0.001			0.097		

[a] OpOpt = optimal habitat dominated only, OptPl = optimal habitat with pine plantations dominated, OptWet = optimal habitat with wet habitats dominated, NoOpt = other, no optimal habitat dominated

[b] number of occurrences in each category

[c] column means followed by the same letter or without letters were not different, $P > 0.05$ (LSD), for % buffer area burned and years since burn, respectively

[d] burns occurred when at least 40% of buffer area was burned

lated (r_s = 0.50, P = 0.082; Table 6). Bobwhite abundance was negatively correlated with April and May PMDI and November rain (r = -0.54, P = 0.055, r = -0.60, P = 0.030, and r = -0.61, P = 0.025, respectively; Figure 1a). Bobwhite abundance was positively correlated with October PMDI and July rain days (r = 0.53, P = 0.061 and r = 0.60, P = 0.030, respectively; Figure 1a).

Bobwhite abundance after DSB was negatively correlated with April and November rain (r = -0.69, P = 0.009 and r = -0.67, P = 0.013, respectively; Figure 1b). Also, bobwhite abundance after DSB was negatively correlated with January, February, winter, and dry season rain days (r = -0.52, P = 0.066, r = -0.59, P = 0.035, r = -0.60, P = 0.032, and r = -0.62, P = 0.024, respectively; Table 7 and Figure 1b).

Bobwhite abundance after GSB was negatively correlated with October, November, December, and Fall PMDI preceding that growing season (r = -0.48, P = 0.099, r = -0.52, P = 0.067, r = -0.50, P = 0.079, and r = -0.52, P = 0.069, respectively; Figure 1c). Also, bobwhite abundance after GSB was negatively correlated with fall rain and December rain days preceding that growing season (r = -0.53, P = 0.061 and r = -0.60, P = 0.030, respectively; Table 7 and Figure 1c). Contrary to normal weather patterns for the region, positive correlations occurred between bobwhite abundance and April rain and January rain days preceding that growing season (r = 0.70, P = 0.008 and r = 0.49, P = 0.087, respectively; Figure 1c).

Discussion

Habitat

Being ground dwelling and nesting birds, bobwhite preferred dry/mesic habitats, especially open

Table 5: Mean bobwhite per station after growing season burns by percent buffer area burned and time since burn for each habitat group at Avon Park Air Force Range, Florida, 1993-2005.

	Habitat group[a]											
	OpOpt			*OptPl*			*OptWet*			*NoOpt*		
	#	SE	n[b]	#	SE	n	#	SE	n	#	SE	n
% buffer area burned, midpoints												
0	1.9	0.12	91	1	0.15	81	1.1B[c]	0.19	58	0.8AB	0.11	119
20	1.9	0.35	26	1.1	0.08	53	0.9B	0.16	16	0.6B	0.1	70
70	2.3	0.16	24	1.4	0.23	24	1.8A	0.31	37	1.1A	0.19	21
P	0.416			0.204			0.034			0.041		
Years since burn[d]												
< 1	2.3A	0.16	24	1.4	0.23	24	1.8	0.31	16	1.1	0.19	21
2	2.5A	0.51	12	1.2	0.23	19	1.6	0.35	18	0.7	0.17	15
3+	0.9B	0.2	18	1	0.43	27				0.8	0.19	48
P	0.009			0.646			0.719			0.478		

[a] OpOpt = optimal habitat dominated only, OptPl = optimal habitat with pine plantations dominated, OptWet = optimal habitat with wet habitats dominated, NoOpt = other, no optimal habitat dominated

[b] number of occurrences in each category

[c] column means followed by the same letter or without letters were not different, $P > 0.05$ (LSD), for % buffer area burned and years since burn, respectively

[d] burns occurred when at least 40% of buffer area was burned

canopy habitats. During the beginning and middle part of the breeding season, drier habitats were not usually flooded by sheet flow; thus, nests were safer from flooding. Open canopy habitats contained fewer perches for avian predators such as hawks and usually had more groundcover vegetation for forage and cover. These habitats also had the necessary microhabitat requirements for the breeding season (i.e., nesting sites with escape and forb cover, bare ground, and food resources nearby), especially if fire was periodically used. Much of the vegetation was heterogeneous across the landscape, with scattered saw palmetto clumps, a diversity of grasses, sedges, and forbs, and scattered low-growing shrub patches. Cattle and other animal trails and localized lower areas created bare ground as well as general scattered bare ground areas. Unlike bobwhites adjusting to the agricultural fields further north, bobwhites in south-central Florida have evolved with vast open habitat that has only recently been fragmented.

As others have shown (DeVos and Mueller 1993, Kitts 2004), pine plantations and closed canopy forests were not preferred habitats because they had a dense canopy, plentiful perches for hawks, less herbaceous vegetation, and much less bare soil. In Mississippi, Lee and Brennan (1994) showed a reduction in the bobwhite population as the forests with >50% canopy cover became more abundant over 38 years in their study area. Using logistic regression and landsat imagery, Schairer et al. (1999) found a negative association between bobwhite abundance and patch size of deciduous forest in Virginia. Provencher et al. (2002) found higher bobwhite abundance after hardwood midstory re-

Table 6: Correlations between number of mean bobwhite per station and annual (Apr-Mar) and seasonal Modified Palmer Drought Severity Index (PMDI), rain, and number of rain days at Avon Park Air Force Range, Florida, 1993-2005.

	PMDI		Rain		Rain days[a]	
Season	*r*	*P*	*r*	*P*	*r*	*P*
Annual	-0.12	0.702	-0.03	0.925	0.23	0.457
Spring	-0.06	0.841	-0.21	0.498	0.02	0.941
Summer	-0.09	0.765	0.30	0.325	0.42	0.143
Fall	0.42	0.153	0.14	0.655	0.24	0.421
Winter	-0.12	0.702	-0.21	0.487	-0.26	0.385
Wet (Jun-Sep)	0.07	0.828	0.41	0.164	0.50	0.082
Dry (Oct-May)	0.01	0.966	0.16	0.603	-0.22	0.470

[a] Rain days = every day that at least 0.25 cm fell

duction treatments over no treatment in northwest Florida sandhills. Cram et al. (2002) had a significant bobwhite abundance response to thinning pine-hardwood stands in Arkansas. Therefore, if the closed canopy were reduced below some level, bobwhite would utilize the area.

Bobwhite showed a higher preference for wet areas in conjunction with dry/mesic habitats than for plantations in conjunction with dry/mesic habitats because some of these wet areas habitats had little or no canopy, were dry during the early breeding season, and had their main food source in south-central Florida, slough grass (*Scleria reticulata*). Slough grass, an annual sedge, was frequently found in wet areas and in disturbed mesic areas. During the study period, many of the hunting season crops had slough grass present as a main component. During the winter, Yoho and Dimmick (1972) and Dixon et al. (1996) had some of their coveys preferentially use wet areas.

Fire

Despite the vegetation having less than 6 months (frequently less than 3 months) to recover after a DSB, bobwhite abundance was elevated in these burned areas the year of the burn as well as the following year. Bobwhite abundance increased more after large acreage fires than small acreage fires. In south Florida, Moore (1972) advocated the use of fire and burning about a third to a half of an area yearly in south Florida pinelands, which supports our findings. In Alabama, Speake (1994) reported increases in fall bobwhite abundance after DSB, particularly with frequent fires. In red-cocked woodpecker (*Picoides borealis*) restoration stands subjected to thinning and fire, Cram et al. (2002) found higher bobwhite abundance after 1 year post burn as compared to unburned and unthinned stands but even higher abundance after 2 years post burn. In south-central Florida, the vegetation can recover quickly from fire and many plant species flower more profusely or only after fire (Myers and Boettcher 1987, Abrahamson and Hartnett 1990, Maliakal et al. 2000, Myers 2000). Increased slough grass seed heads have been found after burns (Frye 1954, Moore 1972). The fires reduced litter and the woody cover for awhile and consumed old saw palmetto fronds, making traveling easier under this frequently used escape and loafing cover and nest sites (Moore 1972). Without fire, saw palmetto and litter cover would increase with time, which would reduce herbaceous and bare ground over time (Maliakal et al. 2000).

An endangered, ground-nesting songbird found

Table 7: Correlations between number of mean bobwhite per station for stations that were burned and annual (Apr-Mar for dormant and Oct-Sept for growing) and seasonal Modified Palmer Drought Severity Index (PMDI), rain, and number of rain days at Avon Park Air Force Range, Florida, 1993-2005.

	Dormant season burns						Growing season burns					
	PMDI		Rain		Rain days[a]		PMDI		Rain		Rain days	
Season	*r*	*P*	*r*	*P*	*r*	*P*	*r*	*P*	*r*	*P*	*r*	*P*
Annual	-0.31	0.299	-0.28	0.348	-0.14	0.646	-0.30	0.316	-0.03	0.935	0.02	0.957
Spring	-0.25	0.415	-0.24	0.425	-0.33	0.273	-0.05	0.859	0.19	0.533	0.25	0.401
Summer	-0.11	0.714	0.01	0.972	0.18	0.566	-0.17	0.569	0.10	0.745	-0.12	0.707
Fall	0.15	0.620	0.09	0.777	0.01	0.982	-0.52	0.069	-0.53	0.061	-0.09	0.761
Winter	-0.42	0.151	-0.47	0.105	-0.60	0.032	-0.36	0.226	-0.10	0.740	0.11	0.726
Wet (Jun-Sep)	-0.02	0.949	0.18	0.565	0.24	0.437	-0.11	0.720	0.16	0.595	0.08	0.789
Dry (Oct-May)	-0.27	0.378	0.04	0.890	-0.62	0.024	-0.29	0.329	-0.30	0.310	0.18	0.556

[a] Rain days = every day that at least 0.25 cm fell

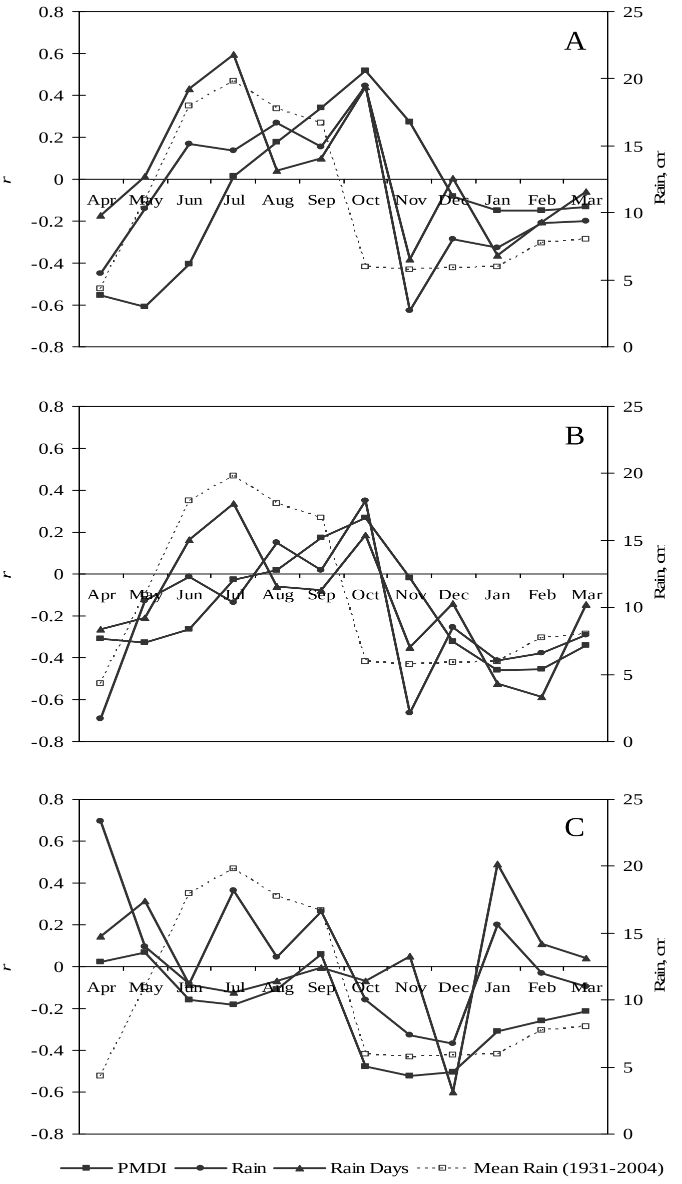

Figure 1: Correlations between number of mean bobwhite per station and monthly Modified Palmer Drought Severity Index (PMDI), rain, and number of rain days at Avon Park Air Force Range, Florida, 1993-2005. A.) Overall abundance. B.) Bobwhite abundance after a dormant season burn bobwhite abundance. C.) Bobwhite abundance after a growing season burn.

in Florida dry prairies, the Florida grasshopper sparrow (*Ammodramus savannarum floridanus*), has also been shown to be positively affected by frequent DSB (Walsh et al. 1995, Delany et al. 2002). Walsh et al. (1995) found that sparrows selected areas recently burned and avoided areas >2 years post burn. Investigating density and reproduction in the sparrows, Delany et al. (2002) reported significantly higher reproductive success at the population level with fire, which reduced with time, but they found no density differences with time since burn. Thus, other ground-nesting birds in dry prairies show similar responses to fire as bobwhite do.

Unfortunately, the GSB burns were not evaluated within a few months of their occurrence because they were mostly conducted during or shortly after each year's call-count. If we had conducted a fall count, differences between GSB with DSB could have been assessed more accurately. Even so, bobwhite abundance did increase within a year of a GSB and did not significantly decline until after 2 years post burn. If the burn was conducted during the drought period in April and May (as most of our burns were), more vegetation would have been consumed with a GSB than a DSB, leaving fewer nesting sites and less escape cover. As with a DSB, vegetation quickly recovers after a GSB, especially once the summer rains start, but shrubs may be more negatively impacted by a GSB. Shriver et al. (1999) found Florida grasshopper sparrows using summer burned areas within a week of their burns and being reproductively active into September. Many of the local plant species flower only or flower more profusely after a GSB, with these mostly being the fall-blooming grasses, sedges, and composites (Abrahamson 1984, Seamon et al. 1989, Robbins and Myers 1992, Streng et al. 1993, Main and Barry 2002). Also, arthropods may increase after burning (Hurst 1972, DeVos and Mueller 1993, Hermann et al. 1998). Therefore, bobwhite abundance almost a year after a GSB may be a reflection of improved forage the preceding summer, fall, and winter before the census.

The main impact after an early GSB would be a reduction of the breeding season due to nest site reduction and increased nest predation, with more nests later in the season than in the early part of the season (Simpson 1972). This is probably why bobwhite abundance peaked when 40 to 59% of a buffer was burned, as there would be areas left unburned. Stations that had optimal habitats and wet areas dominating had a significant increase in bobwhite abundance when greater than 40% of the buffer area was burned, probably due to some areas being too wet to burn and leaving more unburned patches. Weather may ameliorate negative effects of GSB.

Weather

Unlike studies in more arid climates, typical weather patterns were more conducive to bobwhite abundance than deviations from the norm. In the more humid regions of Texas, Bridges et al. (2001) found no correlations between bobwhite abundance and weather. Despite a dry season here, increased wet conditions during the dry season did not have a positive influence on bobwhite breeding abundance.

We did find a positive correlation between bobwhite abundance and July and wet season rain days, but we did not identify an effect due to the amount of rain or PMDI. Though other southeastern weather studies have shown a positive association with a wet or typical summer rainfall and bobwhite abundance or reproduction (Murray 1959, Reid and Goodrum 1960, Speake and Haugen 1960), it seemed more important to have consistent rain rather than large amounts of rain in south-central Florida. Less rain days in July might stress the bobwhite when ambient temperatures are already very high so that more do not survive the harsh summer, especially the young. Increasing rain days might increase the amount of forage on herbaceous plants and insects. Frye (1948) contended that extremely wet summers might actually drown chicks and adults, especially with major flooding events like hurricanes. We found no negative correlations with summer rain, but our study area is generally better drained than that of Frye (1948). He also found higher bobwhite abundance after wet a May, which is contrary to our results. Weather's influence on bobwhite abundance after fire followed the same overall weather trend, except for a positive correlation between bobwhite abundance and April rain and January rain days after a GSB. Fires after a dry fall and winter could result in more fine fuels available for consumption, more vegetation and fuels being consumed and greater of an impact on the resulting vegetation, more wet areas burning that would normally be too wet to burn, and opening up more space (Slocum et al. 2003, Thaxton and Platt 2006). Wet of conditions for DSB would produce very patchy burns and a reduction of effectiveness in increasing usable space if the area was overgrown. With a wet April and more rain days in January, a GSB would probably be more spotty and not burn across the landscape as well (Slocum et al. 2003). Also, the vegetation would probably recover more quickly, increasing usable space more rapidly than in dry years and possibly increasing successful reproduction.

Management Implications

By knowing which habitats in a given area are optimal for bobwhite abundance, managers can better manage for bobwhites. In south-central Florida, optimal habitat was open canopy dry/mesic habitats, especially dry prairie. Reducing canopy coverage and burning increased useable space in these habitats. A mosaic of optimal habitats with marginal or unsuitable habitats is better than no optimal habitat present within the management area. In south-central Florida, bobwhite respond favorably to fire, especially DSB when optimal habitats were present. After 2 years post burn, bobwhite abundance declines. To keep the diverse ground cover intact and viable, the woody cover reduced, and the open ground, frequent fire is needed in these habitats (Moore 1972, Lewis and Harshbarger 1986, Hermann et al. 1998). A burn cycle of 2-4 years would be best in south-central Florida in optimal habitats. Fire effects on bobwhite abundance can interact. A dry winter and dry season produces a better outcome for bobwhite abundance when a DSB is conducted, probably due to more fuel being consumed. Vegetation and insects may colonize quicker after a wet April when a GSB is conducted. As with DSB, GSB benefit bobwhite abundance more after a dry fall and winter. Other interactions of management activities (such as grazing and hunting) with weather should be investigated further to see if management may ameliorate weather impacts on abundance. Also, linking actual reproduction measures and movement studies to these findings would be useful.

Acknowledgments

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Species Visitation at Quail Feeders and Guzzlers in Southern New Mexico

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Providing supplemental feed and water are sometimes used to manage scaled quail (*Callipepla squamata*) in the Chihuahuan Desert even though their biological and economical efficacies are questionable. Seasonal visitation rates of scaled quail and various nontarget species are important parameters affecting the efficacy of feeding and watering practices. However, empirical data on visitation by scaled quail at feeders and guzzlers are lacking. We used video surveillance to assess species visitation at free-choice quail feeders and guzzlers in south-central New Mexico during 2002. Scaled quail accounted for 19.4 and 21.5% of visitations at feeders and guzzlers, respectively. Mourning doves (*Zenaida macroura*), nongame birds, and desert cottontails (*Sylvilagus audubonni*) were the primary nontarget consumers at this site. Relative to similar studies of feeder visitation by northern bobwhite (*Colinus virginianus*) in west Texas, quail feeders tended to be more efficacious (i.e., a greater proportion of the feeder visitations were by quail) in this study. While the biological impacts of feeders and guzzlers remain poorly documented, their use by scaled quail suggests they are important foci within the birds' home ranges. Video surveillance technology permits managers to make data-based decisions on the biological and economic worth of such management efforts. We also describe novel uses for video surveillance relative to facilitating reconnaissance of radiotagged quail whose radios had malfunctioned. Future research should assess the potential for using video surveillance at guzzlers to estimate chick survival in scaled quail.

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Key words: *Callipepla squamata*, Chihuahuan desert, guzzlers, scaled quail, supplemental feeding, surveillance camera, water developments

Introduction

Few management practices have been evaluated to increase abundance of scaled quail (*Callipepla squamata*), whose biological basis for management has lagged notably behind most other North American quails (Rollins 2000). Supplemental feeding (usually with grain, e.g., milo) is a popular, but unproven, management practice for increasing quail (scaled quail, northern bobwhite [*Colinus virginianus*]) abundance on private lands in Texas (Rollins 2007). Most studies suggest that supplemental feeding is only beneficial when natural food sources have become limited (Campbell 1959, Doerr and Silvy 2002, Demaso et al. 2002) and such bottlenecks relative to food availability have not been documented (Guthery 2002, pg. 149). However, other studies have suggested benefits to quail survival and reproduction. Supplemental feeding of northern bobwhite with milo increased overwinter survival during times of severe winter stress in western Oklahoma (Townsend et al. 1999).

There is relatively little information on the efficacy of supplemental feeding for scaled quail. Campbell (1959) reported a modest increase in bird numbers in southeastern New Mexico, but dismissed the efficacy of supplementation as being cost-prohibitive. However, some private landowners may have the capital at their disposal to accommodate supplementation and can control some

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other factors (e.g., hunting pressure) that cannot be controlled on public land. Rollins (2000) reported frequent visitations of adults and young chicks (< 3 weeks old) to feeders in west Texas, and recommended that supplemental feeding be evaluated as a management tool in west Texas.

Water is another management practice often prescribed for quails in arid habitats. Glading (1947) was the first to describe the concept of "gallinaceous guzzlers" (hereafter guzzlers) that have been used commonly since the 1950s to provide drinking water for wildlife in remote arid landscapes. Water developments have benefited some wildlife populations in arid habitats of the western U.S. (Rosenstock et al. 2004) but also have been criticized as potentially rendering the target species more vulnerable to predators (Broyles 1995). Wallmo and Uzzell (1958) and Campbell (1960) summarized their efforts on enhancing scaled quail range with guzzlers and concluded there was no relationship between water availability and scaled quail abundance.

The advent of commercially-available photographic and video surveillance equipment in the past decade has provided a means for non-intrusive assessment of species visitation at quail feeders (Henson 2006) and guzzlers (Rosenstock et al. 2004). Henson (2006) reported that feeder use by bobwhite and scaled quail ranged from 6-16% seasonally at 4 sites in west Texas. Raccoons (*Procyon lotor*), mourning doves (*Zenaida macroura*), and nongame birds were the most frequent visitors in her study.

We initiated a project in 2002 to assess the value of supplemental feeding for increasing survival and recruitment in south-central New Mexico at a site near the northern edge of scaled quail range. A part of this project involved documenting seasonal use of feeders and guzzlers. We used video surveillance of feeders and guzzlers in order to assess their use by scaled quail and nontarget species.

Study Area

Our study area was the privately-owned Armendaris Ranch located 20 km east of Truth or Consequences, Sierra County, New Mexico (Latitude 33.18 N, Longitude 107.03 W). The ranch encompasses some 125,000 ha; our study was in the southeastern quadrant. Most of the ranch resides at elevations between 1375 and 1525 m, and the mean annual precipitation is 25 cm. The primary vegetation is semidesert grassland dominated by black grama (*Bouteloua eriopoda*) and palmilla (*Yucca elata*). Climate is warm-temperate with an average of 213 frost-free days per year (Truth or Consequences, NM). Bison (*Bison bison*) are the predominant grazing animals; other large herbivores include pronghorn (*Antelocaprus americana*), mule deer (*Odocoileus hemionus*), and free-roaming gemsbok (*Oryx gazella*).

Quail feeders and guzzlers were located across the southern half of the ranch. Feeders were constructed from plastic barrels with approximately 12 small (1.3 cm) holes placed at intervals ranging from 2-15 cm from the bottom of the barrel (Figure 1). These barrel feeders were 'free choice' as feed was available at any time throughout the day. Feeders were filled with milo and available year-round; they had been in use for at least 4 years prior to the onset of our study. Feed disappearance through these feeders during the course of this study was estimated at 68 kg/feeder/month (T. E. Wadell, personal communication).

Guzzlers were of several types but typically consisted of a sheet metal rainfall collector that stored water in a fiberglass reservoir directly beneath it (Figure 2). Entrance to a guzzler was restricted to birds and medium-sized mammals by the size of the entrance and the presence of rebar grill covering the entrance. The guzzler's roof was elevated about 50 cm above ground-level, and thus provided a source of shade as well as access to water. Feeder density in our study area was approximately 1 per 40 ha; density of guzzlers was approximately 1 per 300 ha.

Methods

We used motion-sensing video cameras (TrophyView, Inc., Kerens, TX) to assess visitation at feeders during winter (21 Dec - 20 Mar), spring (21 Mar - 20 Jun), and summer (21 Jun - 20 Sep) begin-

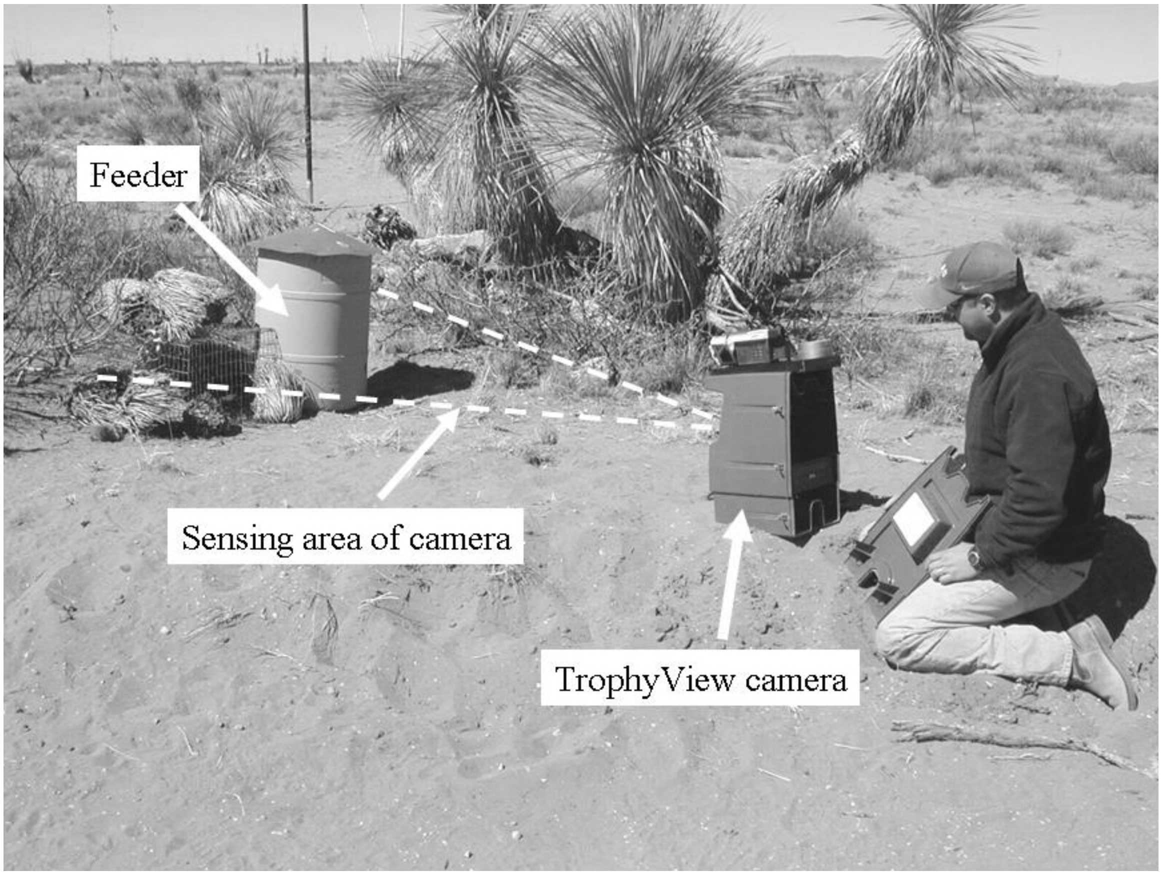

Figure 1: Example of barrel quail feeder used at Armendaris Ranch, Sierra County, New Mexico. TrophyView^{TM}video system is seen in foreground.

ning with winter 2001 and extending through summer 2002, and at guzzlers during spring, and summer seasons in 2002. Two cameras were used and rotated among 4 feeders and 4 guzzlers. These cameras feature a passive-infrared triggering system and provide 24-hour surveillance. Camera systems were placed about 3 m from the feeder or guzzler which provided a horizontal field of view for the camera of approximately 3 m. When triggered, the unit records activity on a VHS videotape. Individual recording events are tagged with date, time, and lunar phase. Recorded tapes were recovered approximately every week and new tapes were inserted. We reviewed videotapes and recorded species, number of individuals, duration at feeder or guzzler (amount of time an individual species spends at feeder or guzzler), time spent actually feeding or watering (as opposed to investigating or shading). A feeding event was recorded if the animal was observed consuming milo. A drinking event was recorded if the animal either walked into the guzzler (and out of site of the camera), or placed its head in the opening of the guzzler. Successive events were not counted as a new event if the same animals returned to the feeder within 30 minutes as a means of ensuring independence among successive observations. Visitation by species was recored by treatment (feeder vs. guzzler) and calculated as a percent of the total number of visitations. We present only descriptive statistics (means and standard errors [SE]) in order to determine 95% confidence intervals. Significant differences were inferred if confidence intervals were non-

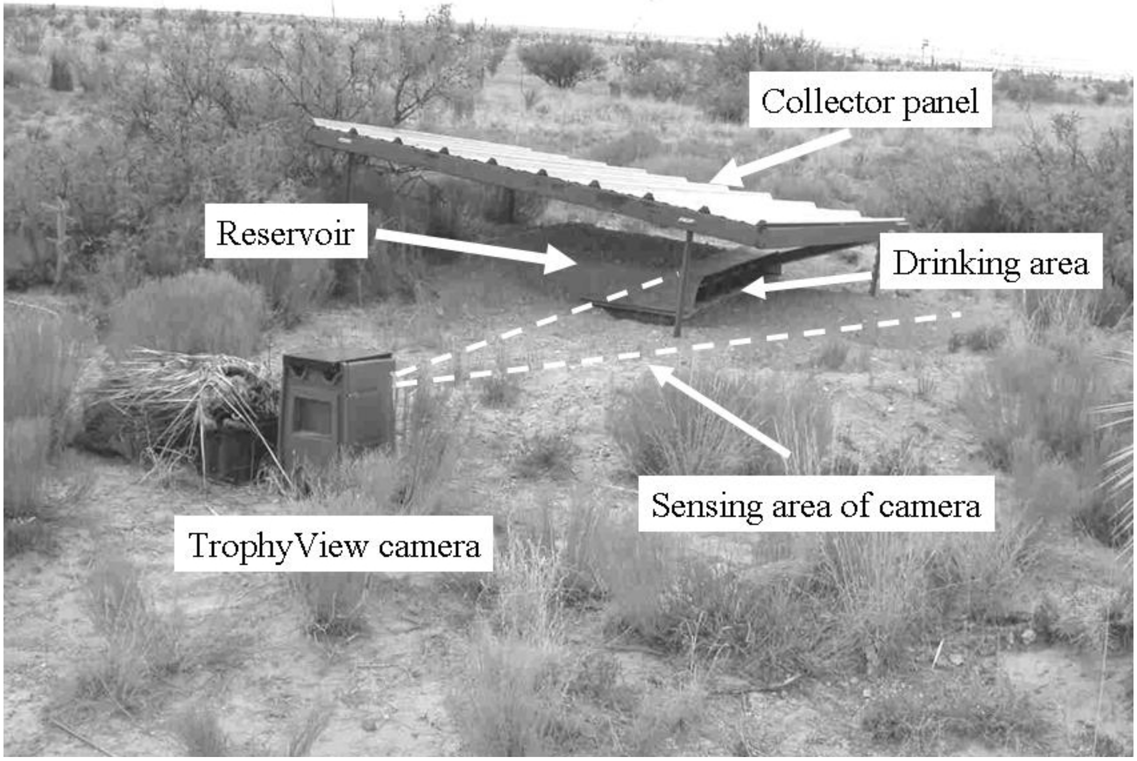

Figure 2: Example of guzzler monitored and placement of TrophyView^{TM}video system, Armendaris Ranch, Sierra County, New Mexico.

overlapping.

Results

We recorded a total of 682 events from a total of 4 feeders and 4 guzzlers that occurred over 150 hours of video surveillance from December 2001-August 2002. We monitored a total of 422 events at feeders and 260 events at guzzlers. Our sample size at guzzlers for winter was only 18 events; these data were not included for subsequent analyses.

Overall, scaled quail represented 19.4% of visitations at feeders and 21.5% of visitations at guzzlers across all seasons (Table 1). Scaled quail comprised a greater proportion of visitations at feeders during winter (43.7%), with less frequent use (i.e., <9%) during spring or summer. The most common species observed at feeders and guzzlers was mourning dove, which accounted for 34.1 and 23.5% of the total visitations, respectively. No mourning doves were recorded in the winter, but they accounted for over half of visitations during spring and summer. Nongame birds accounted for 24.9% and 28.5% of the visitations to feeders and guzzlers, respectively. Feeder visitation by nongame birds was highest during winter. Ravens (*Corvus corax*) were included within nongame birds, and typically accounted for 2-4% of visitations at both feeders and guzzlers. No raptors were videotaped at either feeders or guzzlers. Miscellaneous mammals, which included desert cottontails (*Sylvilagus audubonni*), badger (*Taxidea taxus*), kit fox (*Vulpes macrotis*), bobcat (*Felis rufus*), feral cats, and rodents accounted for 21.6% and 26.5% of visitations at feeders and guzzlers, respectively.

Table 1: Visitations to quail feeders and guzzlers by species as recorded by video surveillance in south-central New Mexico, October-December 2001 and January-August 2002.

Device	Season	Events (*n*)	Scaled quail	Mourning dove	Nongame birds	Misc. mammals
Feeder	Winter	142	62 43.70%	0	53 37.30%	27 19.00%
	Spring	191	17 8.90%	101 52.90%	32 16.80%	41 21.50%
	Summer	89	3 3.40%	43 48.30%	20 22.50%	23 25.80%
	Total	422	82 19.40%	144 34.10%	105 24.90%	91 21.60%
Guzzler	Spring	166	31 18.70%	52 31.30%	44 26.50%	39 23.50%
	Summer	94	25 26.60%	9 9.60%	30 31.90%	30 31.90%
	Total	260	56 21.50%	61 23.50%	74 28.50%	69 26.50%

Scaled quail visitations at feeders and guzzlers averaged 11.2 ± 1.20 min (x ± SE) and 7.6 ± 2.31 min, respectively (Figure 3). Proportion of time spent feeding versus loafing was similar for scaled quail at feeders and guzzlers (34% of time feeding, 65% loafing at feeders; 43% watering, 56% loafing at guzzlers). Scaled quail spent an average of 3.8 ± 0.68 min feeding and 3.3 ± 1.10 min drinking (Figure 3). Mourning dove spent an average of 25.7 ± 4.10 min feeding and 3.8 ± 1.1 min watering. Nongame birds were recorded feeding for an average of 11.1 ± 2.8 min and 1.0 min ± 0.38 min watering.

Discussion

Supplemental Feeding

Proportion of visitations comprised by scaled quail at feeders and guzzlers suggested that both can be viable tools in scaled quail management in this area. Scaled quail were observed at a greater incidence at feeders in southeastern New Mexico than at feeders in west Texas by northern bobwhite (Henson 2006, <10% of visitations across all seasons were by quail [bobwhite and scaled quail] in this study). Accordingly, we suggest that supplemental feeding programs may reap greater benefits to quail in more arid environments. Female scaled quail at this study site with access to free-choice quail feeders survived at greater rates during the breeding season (Mar-Aug) than where feeders were absent (Rollins et al. 2006); access to guzzlers was available at both sites.

Quail visitation at feeders was greatest during the winter in this study similar to the pattern reported by Henson (2006) for bobwhite in west Texas. If feeder visitation by quail is influenced by seasonal energy needs, then this pattern seems logical and

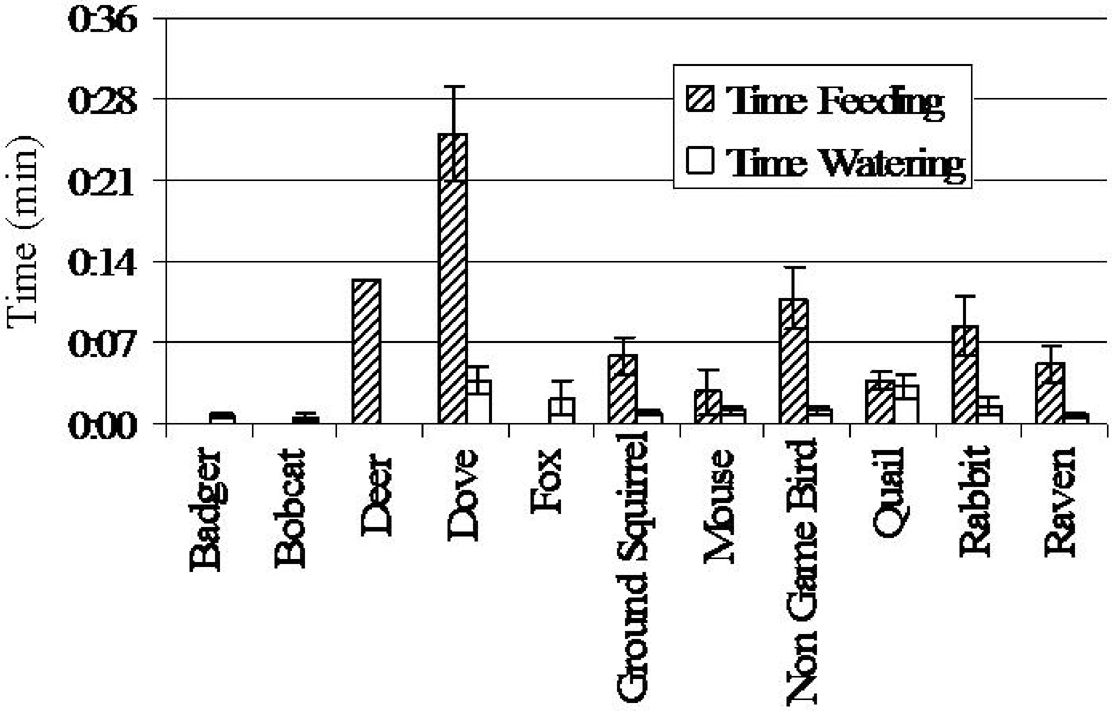

Figure 3: Time spent (minutes) by species at feeders and guzzlers actually feeding or watering in south-central New Mexico.

adaptive. Additional surveillance that includes the fall season, the only season not monitored in our study, is warranted.

The cost of supplemental feeding can be substantial. A conundrum over the efficacy of supplemental feeding of quail remains the issue of what portion of the feed provided is actually consumed by quail as opposed to nontarget species. At the feed disappearance rates estimated on this study site (68 kg/feeder/month), feed costs alone (milo valued at $0.16/kg) would account for $10.58/feeder/month. Expenses for labor and travel are not included in these estimates, but would increase feeding costs substantially. Henson (2006) estimated feeding costs at 4 sites in west Texas at approximately $15/feeder/month for feed costs alone. Raccoons and nongame were responsible for most of the feed lost to nontarget species in her study, and ultimately responsible for inflating the cost of a feeding program at her sites ($15/feeder/month) relative to this study ($10.58/feeder/month). Boyer (1989) reported that an additional quail in the bag could cost from $24 to $60 in successful feeding programs.

Due to the costs associated with a supplemental feeding program, managers should be concerned with the loss of feed to nontarget species at quail feeders. Mourning doves and nongame birds accounted for the majority (59.0%) of visitations at feeders across all seasons. Nontarget species comprised 98% of visitations at quail feeders at 1 site in the Texas panhandle (Guthery et al. 2004). However, as Henson (2006) noted, feeder visitation rates do not necessarily equal feed consumption. Presumably, a small passerine does not consume as much milo per feeding event as does a scaled quail.

Quail biologists have historically dismissed, or demeaned, feeding programs as ineffective, expensive, and even counterproductive. Feeders do concentrate quail and make their locations on the landscape more predictable for hunting purposes

(Rollins 2000, Guthery et al. 2004). Concerns have been expressed that feeders may render quail more vulnerable to various predators, but recent studies (Guthery et al. 2004, Henson 2006) have failed to document such conjecture. Other management concerns regarding supplemental feeding for quail include direct, e.g., potential exposure to aflatoxins (Oberheu and Dabbert 2001), and indirect impacts (attraction of nest-depredating mesomammals [e.g., raccoons] to feeders; Cooper and Ginnett 2000).

The proportion of feeder visitations comprised by scaled quail at this site was 4-10x greater than Guthery et al. (2004) and Henson (2006) recorded for bobwhites in Texas presumably due to more complex assemblages of nontarget species. Raccoons were the predominate species at quail feeders in the Henson (2006) study, accounting for 30-45% of feeder visitations. Raccoons were not present at our study site, and their absence makes feeders more efficacious for scaled quail, and for quail managers in more arid habitats. Henson (2006) concluded that, while feed loss to some nontarget species (e.g., raccoons) may be ameliorated through technology (e.g., electric fencing) or ingenuity, feed loss to other nontarget species (e.g., mourning doves and nongame birds in this study) should be considered an overhead cost of feeding.

Guzzlers

Scaled quail can meet most of their water needs by consuming succulent foods (Hungerford 1960, Schemnitz 1994, Guthery 1999). However, desert quail require drinking water to survive periods of sustained heat and drought (Leopold 1977, pg. 183). Desert quail (*Callipepla* spp., *Oreortyx picta*) frequently drink from catchments, particularly during hot and dry periods (Elder 1956) and show strong fidelity to guzzlers within their home range (Delehanty et al. 2004).

We monitored use of guzzlers during the spring and summer seasons which Rosenstock et al. (2004) identified as the period of peak use of water developments by Gambel's quail and mourning doves in Arizona. Campbell (1960) suggested that water developments were most beneficial for scaled quail in areas characterized by drought during the spring-summer breeding season. Our study area was mired in drought for the duration of the study–annual precipitation in 2001 and 2002 was only 53% and 44% of the long-term means, respectively (Western Regional Climate Center 2003). Above normal temperatures prevailed during this time (e.g., June 2002 had the highest mean monthly temperature on record). Accordingly, our estimates of guzzler use may represent maximum levels. Additional surveillance is warranted to determine guzzler use in cooler, wetter summers and during fall and winter seasons.

Four potentially adverse impacts of water developments in arid habitats are commonly cited: predation, competition, direct mortality, and health problems resulting from poor water quality or disease transmission (Rosenstock et al. 1999). We gathered data during this study that addressed 2 of these concerns (predation and direct mortality).

There is evidence that water developments attract avian and mammalian predators (Cutler 1996, Rosenstock et al. 1999, 2004); however, the effects of these facilities on predator populations and subsequently on scaled quail are unknown. Rosenstock et al. (2004) recorded 4 species of raptors at water developments in Arizona that have been identified as predators of quail (Cooper's hawk [*Accipiter cooperii*], sharp-shinned hawk [*A. striatus*], red-tailed hawk [*Buteo jamaicensis*], and great-horned owl [*Bubo virginianus*]; Rollins and Carroll 2001). However, aside from ravens (a potential egg predator of scaled quail) we did not detect raptors using guzzlers at our site.

The guzzlers used in our study were (by design) fairly specific as to which species could access them, which was documented with low visitations by many nontarget species. The size of the opening on the guzzlers (about 20 cm in height) in our study, and the rebar grill along the opening, precluded larger species of nontarget animals (e.g., mule deer). Guzzlers that feature open drinkers (e.g., like those monitored by Rosenstock et al. 2004), make drinking water available to a wide range of animals-mule deer, turkey vulture (*Cathartes aura*), and coyote constituted the majority of visits to guzzlers. Thus, guzzlers can be successfully designed to limit nontarget use if such is desired (as it was at this study site). Water facilities that facilitate exotic species (in our case free-ranging oryx) should be discouraged (Burkett and Thompson 1994).

We did not detect any direct mortalities (i.e., drowning) in the guzzlers we monitored. Mortalities of birds and small mammals in livestock troughs and other water facilities have been reported (Schemnitz et al. 1998). Rollins et al. (2006) reported that 3 radiotagged scaled quail drowned in the same water trough at the Elephant Mountain Wildlife Management Area in west Texas. The design of the guzzlers used at our site precluded animals from falling into a trough and subsequently drowning, i.e., access to water is on a sloping gradient.

Providing supplemental watering points has been a popular management technique over the past 40 years on many public lands in the American Southwest. Rosenstock et al. (1999) recommended that future wildlife water development projects should: (1) have a solid biological basis, (2) reflect clearly articulated management objectives, and (3) include a formal economic benefit:cost analysis. While we concur with their first and second conclusions, we submit that private land managers are not always constrained by cost:benefit considerations to the same extent expected of public agencies.

Novel Uses For Video Surveillance

We documented novel uses for video surveillance during our study. In December 2002, we experienced massive telemeter failure because of premature battery failure (Rollins et al. 2006). When radiotagged birds disappear, the cause might be emigration or telemeter malfunction. In such cases, considerable time, effort, and expense can be incurred to conduct searches outside the study area (e.g., via aircraft). We employed a TrophyView camera on a feeder in early January, and to our surprise, recorded 3 radiotagged scaled quail on the first day of video surveillance. Subsequent surveillance at other feeders revealed other radiotagged quail (albeit with non-functioning telemeters). Knowledge that ”missing” birds were indeed still present in their former locales allowed us to re-initiate trapping, and subsequently recapture and replace radio on a number of sample birds.

We documented several cases of broods accompanying 1 or both parent quail to guzzlers during the summer. Brood survival is perhaps the most poorly understood aspect of recruitment in quail. We suggest that if parent birds could be marked (perhaps with various combinations of leg bands) and subsequently identified from video surveillance, nonintrusive estimates of chick survival might be obtained. Given that scaled quail used guzzlers much greater than feeders in summer, surveillance of guzzlers would provide the greater probability of repeat videocaptures. Broods of mountain quail exhibited strong fidelity to individual guzzlers in the Mojave Desert (Delehanty et al. 2004), thus surveillance of a particular guzzler could potentially provide numerous videocaptures over time.

Management Implications

Opportunities for active management, e.g., food plots, prescribed burning, are limited for quails in arid and semiarid climates (Rollins 2000). Providing supplemental feed and water for scaled quail has typically been dismissed by quail biologists as either ineffective or too inexpensive. Regardless, feeding is a popular management practice by landowners and hunters, even though it is typically considered a biologically-neutral management practice (Guthery 2002, pg. 149). However, the relatively high level of quail visitations we observed at feeders and guzzlers, and higher survival rates during the breeding season on fed areas (Rollins et al. 2006), indicate that both technologies may have implications for scaled quail management which inhabit more arid habitats than bobwhites. Benefits from providing supplemental feed and water are likely more important for scaled quail under weather conditions similar to those we encountered during our study (below normal precipitation and above normal temperatures).

Providing scaled quail access to feeders and guzzlers likely cannot overcome deficiencies of habitat structure, excessive harvest or other limiting environmental factors, a caveat recognized in bobwhite management (Doerr and Silvy 2002). If habitat structure is inappropriate, then habitat improvement, rather than supplemental feeding-watering, should be the priority of management plans. That said, stationary feeders for scaled quail in the Chihuauhan desert appear to be relatively efficient way to distribute feed.

We acknowledge that the Armendaris Ranch was in better range condition (good to excellent condition) than the adjacent public lands (poor condition) during the drought-characterized period of our study. Thus our ability to assess the absolute impacts of feeding and watering are confounded with range condition.

Costs of constructing guzzlers like those we monitored were estimated at $600 per unit in 2002 (including labor). Given the cost of supplementation efforts, we recommend that managers use some type of surveillance (video or digital photography) in order to make intelligent decisions as to the efficacy of their supplementation efforts. The cost of TrophyView video cameras was $1,100 in 2002; deep cycle marine 12-volt batteries cost about $70, and we suggest at least 2 batteries for each camera system, so that 1 may be recharged and alternated regularly (e.g., every 2 weeks). Less expensive remote cameras (35 mm or digital formats) provide similar estimates of species visitation as video surveillance (Henson 2006) and are considerably less expensive ($200-600/unit) than video surveillance technologies. However we recommend a video system if behaviors of interest (i.e., drinking vs. loafing) need to be quantified.

Access to feeders provided a biological benefit to scaled quail (e.g., increased survival during the breeding season; Rollins et al. 2006) on the Armendaris Ranch, and the concurrent use of guzzlers suggested they can be beneficial to scaled quail. Such benefits likely accrue most during dry years, which are the bane of quail managers. Feeders (especially) and guzzlers are also effective in making quail hunting more predictable and productive. We recognize however that such intensive management practices may not be politically- or cost-feasible on surrounding public lands.

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Population Response of Northern Bobwhite to Field Border Management Practices in Mississippi

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Empirical relationships of the intensity and spatial extent of field border management required to elicit measurable population responses of northern bobwhite are needed. We established 90.5km of herbaceous field borders (6.1 m wide) along row crop field edges on one half of each of 3 - 800-ha agricultural landscapes in northeast Mississippi. Mean percentage of row crop fields established in field borders was 6.0%. During 2000-2002, we measured breeding season abundance and fall density on all 3 sites and survival of radiomarked bobwhite on 2 of the 3 sites. We used space-use models of bobwhite habitat composition and configuration to estimate changes in habitat suitability resulting from field border implementation. Breeding season survival did not differ between bordered (S = 37.2, SE = 0.06) and non-bordered (S = 42.7, SE = 0.09; χ^2_1 = 0.001, P = 0.97) sites. Moreover, bordered and non-bordered sites did not differ significantly with respect to breeding season call counts (bordered = 1.0, SE = 0.18; non-bordered = 0.8, SE = 0.27; $F_{1,10}$ = 0.44, P = 0.22) and fall density (bordered = 0.2 birds/ac, SE = 0.07; non-bordered = 0.1 birds/ac, SE = 0.05; $F_{1,10}$ = 2.18, P = 0.17). However, field borders increased the amount of usable space on average up to 13.1% on bordered landscapes. The relatively low percentage of field borders established on our sites was not sufficient to elicit measurable population responses of bobwhite. We recommend at least 5-10% of a study area be placed in field border habitats to enhance local bobwhite populations.

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Key words: abundance, *Colinus virginianus*, habitat modeling, Mississippi, northern bobwhite, space use, suitability

Introduction

Northern bobwhite (*Colinus virginianus;* hereafter bobwhite) are one of the most intensively studied game-birds in North America; resulting in a wealth of published literature on bobwhite ecology. Despite this wealth of accumulated knowledge, bobwhite populations continue to decline over most of their historic distribution (Church et al. 1993, Sauer et al. 2003). Habitat loss/degradation due to agricultural intensification, conversion of native grasslands to exotic forage grasses, advanced natural succession, intensive grazing, summer mowing, intensive silvicultural practices, and suppression of natural disturbance regimes (fire) have been identified as probable causes for this continued decline (Stoddard 1931, Exum et al. 1982, Roseberry and Klimstra 1984, Brennan 1991, Hunter et al. 2001). Only recently has there been a coordinated range-wide effort to restore bobwhite populations. In 2002, the Southeast Quail Study Group technical committee, as requested by the Directors of the Southeastern Association of Fish and Wildlife Agencies, developed a range-wide bobwhite habitat restoration and population recovery plan.

The Northern Bobwhite Conservation Initiative (NBCI; Dimmick et al. 2002) is a habitat-based conservation plan designed to restore bobwhite populations to levels observed in 1980. One of the primary objectives of NBCI is to increase the amount and enhance the quality of agricultural lands for nesting, brood rearing, and roosting by bobwhites (Dimmick et al. 2002).

Most (81%) of the 2.8 million coveys needed to reach NBCI population goals are predicted to be produced on 8.4 million ha of farmlands within

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the planning region (Dimmick et al. 2002). Creation and maintenance of native grass and forb communities is the primary management practice through which agricultural lands would presumptively be enhanced. Practices encouraged under the USDA National Conservation Buffer Initiative such as field borders, filter strips, and riparian buffers, offer unprecedented opportunities to enhance bobwhite habitat at a scale commensurate with NBCI goals.

Field borders, defined as intentionally managed herbaceous plant communities along crop field edges to provide environmental and wildlife habitat benefits, are one type of conservation buffer practice. Field borders are often used in addition to existing field edge habitats such as fence rows and drainage ditches to reduce soil erosion (Dillaha et al. 1989), increase herbicide and nutrient retention (Webster and Shaw 1996), or to provide wildlife habitat (Davison 1941, Bryan and Best 1991, Puckett et al. 1995, Palmer et al. 2005). Whereas field borders are advocated as a priority in NBCI, the ability of this site-specific management practice to elicit measurable (using currently available indices and census techniques) population responses at local (farm), state, or regional scales throughout the NBCI coverage area is less well understood. Empirical relationships of the intensity and spatial extent of field border management required to elicit measurable population responses of bobwhite are needed. Only Palmer et al. (2005) and Puckett et al. (2000) have examined effects of field border and filter strip habitats, respectively, on bobwhite survival, reproduction, and abundance in the Upper and Lower Coastal Plains of North Carolina.

More than a decade ago, Roseberry (1993) suggested a shift in bobwhite research from emphasis on site and practice-specific research to consideration of the spatial arrangement and extent of habitat management. Likewise, Capel et al. (1993) suggested future research should focus on determining minimal and optimal sizes of habitat management units. However, only Guthery (1997, 1999) has attempted to formalize hypotheses about the quantity and spatial arrangement of suitable habitat as driving mechanisms behind bobwhite population response at local scales. Few experimental studies have addressed this issue. If field borders are to be implemented at a nation-wide scale to increase or maintain bobwhite populations within agricultural landscapes, as encouraged by NBCI and multiple government sponsored programs, empirical relationships regarding intensity and spatial extent of field border practices required to elicit measurable population responses of bobwhite must be addressed. Our objectives were to measure bobwhite population response, survival, and changes in the amount of usable space (Guthery 1997) relative to field border establishment in agricultural landscapes in northeast Mississippi.

Study Area

Our study was conducted on 3 privately owned farms (BRYAN, 3,172 ha; CHANCE, 3,123 ha; MAST, 2,185 ha) in Clay and Lowndes counties, Mississippi. Located within the Black Prairie physiographic region of northeast Mississippi, all sites had a history of agriculture use with most sites actively producing crops for more than 50 years. Primary agricultural practices were row crop (soybean [*Glycine max*], corn [*Zea mays*]), forage, and livestock production. Sites were selected based on similarities in cropping practices, landscape composition (approximately 60-80% row crop), soil associations, and landowner cooperation to maximize homogeneity among and within sites. Most grasslands on each site consisted of perennial, exotic cool (tall fesuce [*Festuca arundinacea*]) and warm season (Bermuda grass [*Cynodon dactylon*], Bahia grass [*Paspalum notatum*]) forage grasses. However, some small remnant and re-introduced stands of native warm-season grasses (big bluestem [*Andropogon gerardii*], little bluestem [*Schizachyrium scoparium*], broomsedge [*Andropodgon* sp.]) were scattered throughout each site. Most linear features (e.g., fencerows, drainage ditches) contained tall fescue and Johnson grass (*Sorghum halapense*). Periodically disturbed areas consisted of early successional grasses and forbs (paspalums

[*Paspalum* sp.], panic grass [*Panicum* sp.], giant ragweed [*Ambrosia trifida*], sumpweed [*Iva annua*], Johnson grass, golden rod (*Solidago* sp.) and partridge pea (*Chamaecrista fasciculata*). Wooded areas were predominantly oak (*Quercus* sp.), green ash (*Fraxinus pennsylvanica*), maple (*Acer* sp.) hickory (*Carya* sp.), sugarberry (*Celtis laevegata*) and eastern red cedar (*Juniperus virginiana*).

During early spring 2000, field borders (6.1 m wide) were established along agriculture field edges (fence rows, drainage ditches, access roads, and contour filter strips) on one half of each site. The field border treatment was assigned randomly to approximately each half of each study site (BRYAN, 1,731 ha; CHANCE 1,471 ha; MAST 1046 ha). For row crop fields receiving field borders, mean field size was 26.9 ha (range = 2.9-146.9; Table 1) and mean percentage of the crop field converted to field borders was 6.0% (range = 0.5-15.3; Table 1). Due to differences in crop field size and configuration among sites, the BRYAN site had a greater percentage of crop field converted to field borders (8.8%) than the CHANCE (4.1%) and MAST (3.0%) sites. We defined our effective site size by buffering all cropping units which received field borders by 800 m (2 x radius of a circular range equivalent in size to the mean home range of resident radiomarked bobwhites). Overall, field borders (54.3 ha) composed between 0.8-1.3% of the total land area of bordered sections of each farm. Row crop field mean shape index was similar among sites (Table 1). Shape index was calculated as the patch perimeter divided by the minimum perimeter possible for a maximally compact patch of the corresponding patch area (McGarigal et al. 2002).

Producers were paid an initial $247.10/ha sign-up bonus with a $123.55/ha/year rental rate paid at the end of each growing season for land planted to field borders. Furthermore, producers were required not to mow, herbicide, or disk field borders during the three years of study. Use of field borders as "turn rows" during harvesting was permitted because this activity occurred generally after the nesting season for most birds, usually involved only one or 2 edges of a field, and facilitated producer participation in the study. Field borders were seeded initially in 2000 with a Kobe lespedeza (*Lespedeza striata*) and partridge pea mix at rates of 11.2 kg/ha and of 3.4 kg/ha, respectively. Following drought conditions during the 2000 growing season which resulted in poor plant growth, field borders were re-seeded in late spring 2001. However, despite these 2 attempts to establish a desired plant community, most field borders re-seeded naturally from seed present within the seed bank (i.e., fallow community). Common species were morningglory (*Ipomea* sp.), crab grass (*Digitaria ciliaris*), Johnson grass, hemp sesbania (*Sesbania exaltata*), nutsedge (*Cyperus esculentes*), and ragweed (*Ambrosia* sp.).

Methods

Survival

Wild bobwhites were captured from January-April each year from 2000-2002 with baited walk-in funnel traps (Stoddard 1931) or by night netting (Truitt and Dailey 2000) on the BRYAN and CHANCE sites. We also captured additional bobwhites from June-July of each year using call-back traps and by night-netting. Captured bobwhite were sexed, aged (adult/sub-adult), weighed, banded with a #7 aluminum leg band, and fitted with a 5-6 g pendant style radio transmitter (American Wildlife Enterprises, Tallahassee, Florida, USA), and then released at the capture site. Radio transmitters operated on 148.000-151.000 MHz bands and were equipped with a 12-hr motion sensitive mortality switch. Capture, handling, tagging, and radiomarking procedures were consistent with the Mississippi State University Institutional Animal Care and Use Committee guidelines (IACUC permit no. #99-212).

We used a programmable scanning receiver (R4000; Advanced Telemetry Systems, Inc., Isanti, MN) with a 3-element Yagi antennae to locate radiomarked birds. Wide-ranging birds were located using fixed wing aircraft. Radiomarked birds were located $\geq$5 times/week during the breeding season (15 April-15 September) by homing to $\leq$40 m and triangulating from positions referenced geographically with a Trimble Geo-Explorer II hand-held global positioning system (GPS) unit. When a mortality signal was detected, we located the transmitter and determined fate of the radiomarked bird using evidence at the recovery site (bird remains, scat, tracks, whitewash) and transmitter damage (Dumke and Pils 1973).

We used Cox's partial likelihood regression (Cox 1975) in PROC PHREG (Allison 1995) in SAS (SAS Institute, Inc. 2002) to estimate survival and test hypotheses of no difference in proportional hazard among genders, ages, years, and treatment effect (bordered/non-bordered). Because of small sample sizes on the CHANCE site, we pooled radiomarked bird failure times with those of the BRYAN site. Radiomarked birds which moved $\geq$2 km from the sites prior to the breeding season were excluded from analyses. Birds which moved off the site during the breeding season were right censored on the last date they were found on the site. Likewise, birds for which transmitter failure was suspected were censored on the last date a transmitter signal was recorded. Breeding season survival was based on a 154-day interval (15 April-15 September) beginning with covey breakup and initiation of reproduction (Burger et al. 1995*b*) except for the non-bordered section of the CHANCE site in 2000. All radiomarked birds (n = 4; 3 female, 1 male) captured during the winter trapping session on this area died by early May before additional birds could be captured and radiomarked. Therefore, we only report survival estimates for the period of 23 May-15 September for the non-bordered sections of the BRYAN and CHANCE sites for 2000 only. This survival estimate does not reflect survival for the entire breeding interval (15 April - 15 September). We assumed gender and age classes were sampled randomly, individual survival times were independent, the censoring mechanism was random, and capturing, handling, and radiomarking did not affect survival (Pollock et al. 1989). Results of all tests were considered significant at α = 0.05.

Population response

We estimated annual fall density and relative covey abundance of bobwhites using the fall covey-call technique (Seiler et al. 2002, Wellendorf et al. 2004). We defined our sampling frame by overlaying a grid composed of 500-m x 500-m cells on each site to identify the pool of potential sampling cells within the ownership boundaries. For each site, we then selected randomly 3 cells from each of the bordered and non-bordered sections. Cell selections were re-randomized each year. Covey counts were conducted during late October-early November 2000-2002 (Wellendorf et al. 2002, 2004).

We placed one observer at midpoints along each side of a sampling cell $>$0.5 hours before sunrise (CST) to monitor morning covey calls until 0.25 hours after sunrise. Observers recorded time, azimuth, duration, and number of covey calls per calling event for coveys within and outside the cell. We then triangulated covey locations based on observer azimuths plotted on 1:10,000 scale GIS land cover/land use maps in relation to time of calling activity to determine number of coveys within the sampling cell. Because of likely differences among observers' abilities (hearing acuity, experience, and attentiveness) to detect covey calls, we used only covey detections for which $>$2 observers recorded a calling event at approximately the same time and location for fall density estimation. However, when estimating relative covey abundance (coveys heard/observer/morning), we used all covey detections recorded per observer regardless of the number of other observers who may have recorded the same calling event and covey location (within and outside of the sampling cell). This approach requires several assumptions because multiple observers may have detected the same calling event. However, we believe this approach to be a valid index of relative covey abundance because cells were distributed randomly and because effort (observers/cell) and intensity (# of cells/farm section) remained constant throughout the study. For a detailed description of density, calling rate, and variance estimation procedures see Smith (2001, 2004).

Table 1: Number, percentage of field border per field, size (ha), and shape index of fields receiving field borders at BRYAN, CHANCE, and MAST sites in Clay and Lowndes counties, MS, 2000-2002.

		% Field Border		Field Size		Shape Index	
Site	n	$\bar{x}$	SE	$\bar{x}$	SE	$\bar{x}$	SE
BRYAN	16	8.8	0.89	19.6	5.6	1.6	0.12
CHANCE	18	4.1	0.61	18.8	3.3	2.2	0.25
MAST	3	3.0	0.90	114.8	20.1	1.9	0.59
Overall	37	6.0	0.63	26.9	5.4	1.8	0.15

We used breeding season call counts (Bennitt 1951) to index annual bobwhite breeding density. Stations were arranged in grid fashion with a 800-m spacing between stations. All stations were geo-referenced and the same set of stations was used throughout the study. Call counts were conducted in mid-June between 0545-0900 hrs (CST) with average wind speeds <15mph. We recorded number of calling males heard during a 5-minute listening period at 102 stations (55 bordered, 47 non-bordered). Counts were conducted twice at each station during a 4-day interval each year.

For fall and breeding season relative abundance measures, we used a repeated measures ANOVA in a randomized complete block design in PROC MIXED (SAS Institute, Inc. 2002) to test the null hypothesis that abundance measures did not differ between bordered and non-bordered sites during the 3 years of study. Because subtle differences in landscape context and farming practices existed among the sites, we used site as a random block effect whereas year was the repeated time effect. The annual population measure at the half-site was used as the response variable. We modeled within-subject covariance (i.e., the repeated year effect) using the autoregressive (order 1) covariance structure. Results of all tests were considered significant at α = 0.05.

Habitat Modeling

We created a Geographic Information System (GIS) coverage for each site by digitizing land cover/land use polygons from 1:12,000 georeferenced Digital Orthophoto Quarter Quadrangle (DOQQ) imagery. We used a hand held Global Positioning Systems (GPS) unit or IKONOS multispectral imagery acquired during the study for updating coverages. We subjectively grouped land cover/land use classes into habitat classes based on management regimes and/or similarities in vegetation characteristics. Pasture/hay fields, CRP fields, cover strips, and filter strips (GRASS) were grouped together due to similarity in structural characteristics, species composition, and/or lack of periodic disturbance. Wood lots, wooded fence rows and ditches, and road right of ways containing woody vegetation were grouped as WOODY. Row crops (ROWCROP) included soybeans, corn, or food plots. Roads, buildings, and water were classified as ”ODD” areas. Field borders were classified as early succession habitats (SUCC). We created 2 GIS habitat coverages for the bordered sections of each site; one coverage before field borders were established and one coverage after field borders were established. We then converted these vector-based coverages to grid coverages (10m cell size) for analyses. To measure the impact of field borders on bobwhite habitat suitability, we used a logistic regression-based

Table 2: Breeding season (15 April-15 September) survival of radiomarked northern bobwhite at field border management sites in Clay and Lowndes counties, MS, 2000-2002.

	Border			No Border			Overall		
Year	n	S[a]	SE[b]	n	S[a]	SE[b]	n	S[a]	SE[b]
2000	37	34.3	0.10	16	75.0[c]	0.13	53	43.3	0.09
2001	37	35.6	0.09	34	32.1	0.09	71	33.1	0.07
2002	21	41	0.12	19	54.6	0.16	40	50.6	0.09
Pooled	95	37.2	0.06	69	42.7	0.09	164	40.8	0.05

[a] Interval survival

[b] Standard error of survival

[c] Survival from 23 May - 15 September

space-use model constructed from differences in the composition and structure of habitat patches within the home ranges of 285 radio-marked bobwhite and those of random ranges in the same landscape context as this study (Smith 2004). This model contained habitat metrics describing the relative dominance of woody patches (woody_lpi), percentage of early succession habitat (succ_pland), and total perimeter-to-area ratio of all patches (paramn). See McGarigal et al. (2002) for a description of metric formulae. We used the posterior probabilities generated from this logistic regression model to measure and spatially map bobwhite habitat suitability for the bordered sections of each site (HSI; Brennan et al. 1986). HSI was computed as:

$$\frac{e^{(-1.4916-0.0529(woody\_lpi)+0.0456(suc\_pland)+0.0011(paramn))}}{1+e^{(-1.4916-0.0529(woody\_lpi)+0.0456(suc\_pland)+0.0011(paramn))}}$$

We used the moving window process with a 400-m search radius in FRAGSTATS (McGarigal et al. 2002) to calculate respective model metrics for each grid cell in the land cover grid. We used a 400-m search radius because this distance produced an area equivalent in size to the median home range of bobwhites in this study. The value of the habitat metric generated within this 400-meter window was assigned to the center grid cell within the window. This process was repeated for each grid cell within the land cover grid, resulting in one habitat metric grid for each habitat metric contained within a model. We then used these habitat metric grids to computed respective HSI grids (posterior classification probabilities) using map algebra functions in ARCINFO GRID. We classified HSI values into 4 categories representing excellent (1.0-0.9), good (0.9-0.75), fair (0.75-0.5), and unsuitable (<0.5) habitat and report percentage change in each HSI category before and after border establishment for each site. More specific details on habitat model development are reported in Smith (2004).

Results

Survival

During 2000-2002, we radiomarked 209 bobwhite. However, only 168 birds were alive during the breeding season (15 April-15 September). Of these, 98 birds were right censored due to survival past the end of the breeding season (n = 49), movement from the site or loss of signal (n = 44), and transmitter failure or accidental researcher induced mortality (n = 5). Primary sources of mortality included avian (n = 6), mammalian (n = 20), unknown predator (n = 41), and unknown cause of death (n

= 3). Unknown mortalities were events in which an intact bird was found but no identifiable source of mortality could be identified. All intact birds were decomposed to an extent to preclude necropsy.

Breeding season survival did not differ by sex (MALE = 42.8, SE = 0.06, FEMALE = 31.3, SE = 0.08; χ^2_1 = 0.409, P = 0.52), age (ADULT = 30.7, SE = 0.09, SUB-ADULT = 41.0, SE = 0.05; χ^2_1 = 0.443, P = 0.44), or among years (2000 = 43.7, SE = 0.09, 2001 = 33.1, SE = 0.07, 2002 = 50.6, SE = 0.09; χ^2_1 = 0.428, P = 0.51). Furthermore, breeding season survival did not differ between bordered (S = 37.2, SE = 0.06) and non-bordered (S = 42.7, SE = 0.09; χ^2_1 = 1.707, P = 0.191; Table 2) sites. Overall breeding season survival was 40.8% (SE = 0.05; Table 2).

Population response

Fall density (birds/acre) did not differ between bordered ($\bar{\chi}$ = 0.18, SE = 0.067) and non-bordered ($\bar{\chi}$ = 0.11, SE = 0.049) sites ($F_{1,10}$ = 2.18, P = 0.17; Table 3). Likewise, number of coveys detected/observer/morning did not differ between bordered ($\bar{\chi}$ = 0.71, SE = 0.228) and non-bordered ($\bar{\chi}$ = 0.46, SE = 0.149) sites ($F_{1,10}$ = 3.34, P = 0.10; Table 3). Mean number of calling males/station during the breeding season did not differ between bordered ($\bar{\chi}$ = 0.98, SE = 0.181) and non-bordered ($\bar{\chi}$ = 0.80, SE = 0.269) sites ($F_{1,10}$ = 0.44, P = 0.22; Table 3).

Habitat Modeling

Overall landscape suitability was greater after field border establishment for the BRYAN (HSI_{before} = 0.32; HSI_{after} = 0.36), CHANCE (HSI_{before} = 0.23; HSI_{after} = 0.28), and MAST (HSI_{before} = 0.38; HSI_{after} = 0.49) sites. Assuming HSI $>$ 0.50 represents suitable habitat, habitat suitability increased by 7.6% on the BRYAN, 7.8% on the CHANCE, and 23.9% on the MAST sites (Table 4). However, relative effect size [(Bordered number of pixels with HSI $>$ 0.5 - Non-bordered number of pixels with HSI $>$ 0.5 for non-bordered) / (Non-bordered number of pixels with HSI $>$ 0.5)] differed among the study sites. Relative effect of field border establishment was greatest for the MAST site (79.6%), intermediate for the CHANCE site (66.7%), and lowest for the BRYAN site (33.9%). Field borders did not affect the amount excellent (HSI = 1.0-0.90) habitat on the CHANCE and MAST sites and only slightly (0.3%) increased the amount of excellent habitat on the BRYAN site.

Discussion

Survival is a critical component governing bobwhite population growth. Bobwhites in our study experienced similar survival to those reported in other studies within agricultural landscapes (33.2% Burger et al. 1995*a*), but lesser survival than on intensively managed areas (43.8%, 46.9%, 50.9%, respectively; Burger et al. 1998, Smith 2001, Taylor et al. 2000). Adult (30.7%) and sub-adult (41.0%) survival of radiomarked bobwhite in our study were similar to those reported in Puckett et al. (1995) (adult = 28%, sub-adult = 41%). Management techniques (e.g., burning, disking) recommended by Stoddard (1931), Rosene (1969) and others are practiced today to elicit positive population responses. Presumptively, these responses stem from increases in population vital rates (survival, reproduction) or rates of immigration. However, identifying and understanding the specific fitness benefits of field border management practices has been more difficult. Although the point estimates of survival for bordered areas suggest that bobwhite inhabiting field border areas may have experienced greater survival during the 2001 breeding season, this difference was not sufficient to elicit a measurable response in abundance during Fall 2001.

The NBCI is predicated on the assumption that nesting and brood-rearing habitat is lacking in agricultural landscapes and addition of native warm-season grass and forb communities will provide this essential resource (Dimmick et al. 2002). The presumption is that population response will occur through increased reproductive effort and/or success associated with expanded breeding habitat. Puckett et al. (2000) and Palmer et al. (2005) attributed population responses observed in their studies to enhanced reproductive success. The modest population response we observed clearly did

Table 3: Overall mean response of fall bird density (birds/acre), coveys detected/observer, and breeding season call counts for field border management sites in Clay, Lowndes, and Noxubee counties, MS, 1999-2002.

	Border		No Border		
	$\bar{x}$	SE	$\bar{x}$	SE	P-value
Fall Density	0.18	0.067	0.11	0.049	0.171
Coveys Heard	0.71	0.228	0.46	0.149	0.097
Call Count	0.98	0.181	0.80	0.269	0.219

not occur through substantively enhanced survival, thus increased reproductive effort or success was the likely mechanism for response. Although we collected information regarding reproductive performance, insufficient numbers of nests were available to obtain reliable estimates of reproductive success, thus precluding definitive statements regarding the role of reproduction in our results.

Bobwhite exhibit substantial reproductive ability to respond dramatically to favorable habitat conditions. Puckett et al. (2000) reported 59.1% more calling males/stations on one of 2 sites where filter strips were established. Greater abundance was recorded on both sites for breeding season flush count (430%) and catch-per-effort (89.3%) indices. Similarly, Palmer et al. (2005) observed on average 36.1% more bobwhite coveys on sites with field borders and predator removal than non-bordered sites across 3 years of study. Field borders in Palmer et al. (2005) comprised between 9-13% of the tillable land across all study sites. We observed similar, although not significant, relative effect sizes in fall density (65.8%), coveys heard/observer/morning (55.7%), and breeding season call counts (23.3%) indices to those reported in Puckett et al. (2000) and Palmer et al. (2005). Our observed effect sizes with a smaller amount of tillable land converted to field border habitats (6.0%) suggest that field borders may enhance bobwhite abundance, but given the magnitude of variation of the estimates and relatively small number of replicate landscapes in our study, these differences were not detectable statistically.

Field borders in our study were of comparable width (6.1 m) to the filter strips (6.9 m) in Puckett et al. (2000) and field borders (3.0-5.0 m) in Palmer et al. (2005). However, field borders in Puckett et al. (2000) comprised 4.9-9.4% of the landscape and approximately 5.6% of the Wilson county site. Field borders in our study comprised only 0.8-1.3% of the landscape of bordered sites. Because study area boundaries of the above studies were defined by land ownership without regard to the species under study, quantifying the percentage of field border habitats depends on the balance of other habitats within the study area. We defined our effective site size by buffering all cropping units which received field borders by 800 m (2x mean home range size of resident radiomarked bobwhites) which may differ from methods used in Puckett et al. (2000) for delineating study area boundaries, thus influencing percentage of land area in field borders. Based upon percentage of land area in field borders and field border width, the study area in Puckett et al. (2000) was more complex (i.e., greater edge density of field borders/ha) than sites in our study. Similarly, the size of fields receiving field borders in our study (26.9 ha) was much greater than those in Wilson (1.8 ha) and Hyde and Terrell (8 ha) counties in Palmer et al.

Table 4: Change in habitat suitability index values resulting from field border establishment on treatment sites in Clay, Lowndes, and Noxubee counties, MS, 1999-2002.

	BRYAN		CHANCE		MAST	
	Border	No Border	Border	No Border	Border	No Border
HSI	% [a]	%	%	%	%	%
1.000.90	0.6	0.3	0.0	0.0	0.0	0.0
0.900.75	9.5	4.6	8.5	3.7	21.9	8.4
0.750.50	19.9	17.5	11.0	8.0	32.0	21.6
<0.50	70.0	77.6	80.5	88.3	46.1	70.0

[a] Percentage of study area

(2005). In the context of Puckett et al. (2000) and Palmer et al. (2005), our results suggest that the percentage of the land base established in field borders, and presumably usable space, may play a greater role in eliciting population responses of bobwhite than field border width.

Although field borders have been demonstrated to elicit population responses in other studies (Puckett et al. 2000, Palmer et al. 2005), we observed only qualitative, not statistical, differences in abundance when field borders comprise between 0.8-1.3% of the land area of bordered sections of each site. Therefore, given our results in the context of those reported in Puckett et al. (1995, 2000) and Palmer et al. (2005), we suggest that at least 5-10% of a site be placed in field border habitats to elicit measurable responses from bobwhite populations.

On average, field borders increased the amount of suitable habitat for bobwhites by 13.1% across all sites with a mean relative effect size of 60.1%. However, overall mean landscape suitability remained relatively low (HSI = 0.23-0.49). This was due primarily to the relatively poor habitat that remained over the balance of the study sites. Change in habitat suitability was most pronounced on the MAST site which had substantially greater field size, thus less percentage in field borders than the BRYAN or CHANCE sites. This site was typical of most large scale production systems which emphasize field consolidation. Establishment of field borders within these systems will have a greater net effect on whole-farm habitat suitability for bobwhites. Field borders were not able to alter the amount of area deemed excellent (HSI = 1.0-0.90) on 2 of 3 sites; suggesting that field borders alone will not significantly provide optimum habitat across an entire farm. Insofar as a 0.8-1.3% change in land use resulted in a 7.6-23.9% increase in the amount of suitable bobwhite habitat, additional management (i.e., prescribed fire, herbicide renovation, etc.) of surrounding habitats will be necessary to further elevate whole-farm habitat suitability.

Resource professionals are being held under greater scrutiny and accountability for resources (time, money, etc.) expended on wildlife conservation. The NBCI plan estimated that changes in land management on 33.1 million ha of farm, forest, and range land would be necessary to achieve population goals; however, primary land use would be affected on only 6.2% of this area. Given that 5-10% of a farm must be converted to field border practices to elicit a measurable bobwhite response, additional management of surrounding habitats may be required to further elevate and maintain bobwhite populations above pre-management levels (Palmer et al. 2005). Field border management practices encouraged by NBCI and the USDA National Conservation Buffer Initiative can be used to enhance bobwhite populations. However, the amount of field borders established will likely govern their ability to evoke measurable changes in population parameters or abundance. USDA conservation practices, such as the recently announced CP-33 (Bobwhite Buffers) practice, may enhance bobwhite habitat in agricultural landscapes with minimal changes in primary land use only if sufficient acreage is established as field border habitat. Using a focal area approach to target delivery of conservation practices such as CP33 may help to increase the proportion of the local landscape impacted above thresholds required to elicit measurable bobwhite population responses.

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Northern Bobwhite Demographic and Population Response Following an Intensive Habitat Modification to an Agricultural Landscape

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Northern Bobwhite (*Colinus virginianus*) populations have been declining throughout most of their endemic range due to numerous factors (e.g., increased urbanization, predators); however, changing land-use practices have proved most detrimental to bobwhites. In parts of the southeastern USA, small-scale farming has been replaced by large-scale center-pivot irrigated fields and this has exacerbated habitat loss. Despite these trends, bobwhite populations in the Southeast have remained stable or increased on many areas employing intensive habitat management regimes, substantiating the importance of appropriate habitat management for long-term bobwhite persistence. In effort to reverse one such decline, we intensively modified a center-pivot, agriculture dominated landscape to benefit bobwhites by creating new habitat and improving existing habitat. Techniques utilized to modify this landscape were: establishment of linear habitats (field borders and buffer strips); planting longleaf pines; and management of existing habitat via prescribed burning and timber management. During 1998-2001, we monitored bobwhite (n = 498) demographics and population response following annual habitat restoration and management using radio-telemetry and fall abundance estimation (i.e., covey call-counts). Average survival during over-winter (0.4698, SE = 0.0721), breeding (0.3561, SE = 0 .0667) and annual (0.1673, SE = 0.0411) time-periods were higher than those reported for other agriculture studies and similar to those of intensively managed, "plantation" habitats. Bobwhite coveys and broods used newly developed longleaf pine, linear habitats (e.g. field borders/hedgerows), and managed woodlands. Further, nest site selection was commonly associated with these novel habitat types. As a result of the positive demographic response to habitat modification, bobwhite abundance also improved during the study. Consequently, we surmised that modification of agricultural landscapes may improve habitat quality and quantity for bobwhites and subsequently help to increase demographic rates and bobwhite abundance.

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Key words: agriculture, buffer strips, field borders, habitat use, home range, northern bobwhite, PROC NLMIXED, Program MARK, reproduction, SAS, survival

Introduction

Despite being the most studied upland gamebird in North America, northern bobwhite (*Colinus virginianus*) populations have continued to decline throughout most of their endemic range. Whereas declining populations have been associated with various factors (e.g., increased urbanization, changing predator dynamics), changing land-use practices have proved most detrimental to bobwhites (Brennan 1991, 1999, Church et al. 1993, Rollins and Carroll 2001). Recent changes among agriculture landscapes have dramatically affected bobwhites by reducing habitat quantity and quality (Brennan 1999). Clean farming, larger fields, center-pivot irrigation systems and increased herbicide and pesticide use (Capel et al. 1993, Sotherton et al. 1993) have become a common rubric among these landscapes-an ecosystem which once supported high densities of bobwhites. As these habitats, which once benefited

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bobwhites, have diminished, bobwhite populations have also waned. Further, more intensive or lack of management among remaining forested habitats (e.g. silviculture) surrounding agriculture fields has also contributed to habitat loss (Burger 2002). Notably, the declining status of bobwhite populations are not unique to bobwhites, but have also affected numerous species of songbirds (Conover 2005). Contrary to these trends, bobwhite populations in the Southeast have not declined on many areas that have employed intensive habitat management regimes (Brennan et al. 2000, Palmer et al. 2002, Stribling and Sisson 2009); this substantiates the importance of appropriate habitat management to maintaining long-term bobwhite populations.

During the past decade, in an attempt to mitigate habitat loss among agricultural landscapes, federal Farm Bill programs (e.g., CRP, WHIP, EQIP) have been implemented to provide landowners monetary incentive to restore or set aside portions of their cropland to promote early-succession vegetation (Burger 2002). Numerous management practices qualify for enrollment in these programs benefiting bobwhites and other species: cool- and warm-season grass plantings; conservation tillage; exotic grass control; wildlife habitat improvement or development; pine tree management; and linear habitats (LH) including filter or buffer strips, field borders and riparian buffers. However, the utility of these linear habitats, pine plantings, and other habitats, when applied to agricultural ecosystems, to bobwhite demographics and population growth is poorly understood.

Previous research has demonstrated that bobwhites used, and in some cases preferred, LHs for normal daily activities, brood-rearing and nesting (Puckett et al. 2000, Cook 2004). Likewise, previous research has documented increases in bobwhite and songbird abundance when combined with mesomammal reduction (Bromley et al. 2000) and without mesomammal reduction (Hamrick 2002, Cook 2004, Conover 2005). Additionally, Cook (2004) found that bobwhites on areas with LHs exhibited higher survival and lower dispersal proclivities compared to areas without LHs.

However, despite the wide-spread habitat implementation gained from Farm Bill programs and the purported population increase associated with LHs and agricultural ecosystems as mentioned above, the utility of LHs relative to bobwhite demographic parameters at both the local and regional scale remains uncertain. Similarly, few studies have examined the utility of planting longleaf pines (PPs) among agriculture landscapes to improve bobwhite habitat. Whereas previous studies revealed that bobwhite abundance increased on areas with LHs compared to areas without them (Bromley et al. 2000, Hamrick 2002), their methods employed could not provide the means to adequately ascribe whether the observed population increase was a consequence of higher survival, increased reproductive success, or due to immigration. Furthermore, although Puckett et al. (2000) suggested that linear habitats were preferred among bobwhites, they reported that nest survival was low, particularly during the early nesting season. Moreover, Cook (2004) suggested that more research was needed to examine the utility of linear habitats to bobwhite broods. Thus, more research has been warranted to ascertain whether novel habitats improve demographic parameters and provide a practical utility to facilitate reversal of population declines observed among agricultural landscapes.

The primary objective of this study was to examine the utility of augmenting an agricultural dominated landscape with novel habitat types and examine bobwhite habitat-use, demographics, and population response following an intensive modification. We intensively modified the center-pivot, agriculture dominated landscape to benefit bobwhites by creating new habitat and improving existing habitat. Techniques utilized to modify this landscape were: establishment of linear habitats, field borders and buffer strips; planting longleaf pines at a conservative spacing; and management of existing habitats via prescribed burning and timber management. Finally, we compared our results, when applicable, from this study to intensively managed sites and an unmanaged agriculture site in southwest Georgia since we did not have pre-treatment demographic data.

Study Area

The study was conducted on a privately-owned property, Whitehall Plantation (3734 ha), in Laurens and Bleckley counties, Georgia, USA. This study site was located in the Upper Coastal Plain physiographic region near the fall line. Prior to intensive habitat modification during 1998-1999, the study site was comprised of dry and irrigated agriculture fields (55%), unmanaged woodlands (40%) comprised of mixed hardwoods and pines (*Pinus* spp.), and 5% other, miscellaneous-type habitats (e.g., pastures, ponds). During this time, the primary land-use objective was agriculture (i.e., row-crop farming) and the estimated bobwhite population was <1 bird/4 ha. However, during 1997, the primary land-use objective changed to management that benefited northern bobwhites, but farming remained an objective-albeit secondary.

During 1998-1999, intensive habitat management was undertaken converting the agriculture predominated landscape to a landscape more conducive to bobwhites. We employed numerous habitat techniques to improve habitat for bobwhites: dry-land agriculture fields were planted in longleaf pines (*Pinus palustris*); 15 m field borders, buffer strips, and hedgerows were created in all irrigated, agriculture fields; no-tillage farming practices was implemented; annual autumn disking and fallow field management was employed to stimulate annual weed production and arthropods for bobwhite broods; and both chemical and mechanical silvicultural treatments to decrease basal area (timber density) among upland and lowland timberland areas was applied as needed. As such, the new landscape matrix was comprised of agriculture (22%), managed woodlands (21%), and planted longleaf (21%) with interspersed linear habitats (LH [12%]; hedgerows, terraces and field borders), hardwoods (10%), other (ponds, pastures, etc; 8%) and fallow fields (6%).

Methods

Trapping and Monitoring

We trapped bobwhites during October-November and March-April 1998 - 2002 using standard funnel traps (Stoddard 1931) baited with grain sorghum and cracked corn. We covered traps with brush (e.g., fresh-cut pine limbs) to minimize stress on captured birds and to conceal traps from predators. We classified bobwhites by age and gender, and we weighed, leg-banded and released them at the capture sites. We outfitted birds weighing ≥132 g with pendant-style (Mueller et al. 1988) transmitters (6.0 g) equipped with an activity switch (Holohil Systems Ltd., Ontario, Canada). Trapping, handling, and marking procedures were consistent with the guidelines in the American Ornithologists' Union Report of Committee on the Use of Wild Birds in Research (American Ornithologists' Union 1988) and the protocol was approved by the Auburn University Institutional Animal Care and Use Committee, IACUC (Protocol Review Numbers: 2002-0364).

Survival - We monitored bobwhites ≥3 times weekly using the homing method (White and Garrott 1990, pg. 42). We approached birds within 25-50 m to minimize location and classification errors; and entered the locations into a geo-database using Geographic Information Systems (GIS) and ArcView® software (Environmental Systems Research Institute, Inc.). We determined specific causes of mortality when possible, by evidence at the kill site and condition of the radio-transmitter (Curtis et al. 1988). When radio contact was lost, we systematically searched on and off the study area within approximately 5 km of the bird's last known location.

Reproduction - During nesting season, we assumed inactive birds, determined via an activity switch, observed in the same location on 2 consecutive days to be nesting. We approached inactive hens and marked their location with flagging tape at a distance of 5-10 m and recorded the location in our geo-database. We monitored nests ≥5 times weekly and determined exact nest location and number of eggs when the incubating hen left the nest to feed.

Table 1: Models explaining northern bobwhite survival derived via Program MARK (known-fate model; c-hat = 1.78) relative to gender- and time-dependent factors for Whitehall Plantation located in Laurens and Bleckley County, Georgia, 1999 - 2002.

Model	K	$QAIC_c$	$\Delta\, QAIC_c$	QDeviance	W_i
S(season-constant)	2	1605.8567	0.0000	1601.8554	0.4700
S(season + gender)	3	1607.4971	1.6404	1601.4942	0.2070
S(.)	1	1608.2812	2.4245	1606.2806	0.1398
S(season+gender*season interaction)	4	1608.9472	3.0905	1600.9428	0.1002
S(. + gender)	2	1609.7107	3.8540	1605.7095	0.0684
S(annual-constant)	4	1613.6336	7.7769	1605.6295	0.0096
S(season-time)	7	1614.9662	9.1095	1600.9549	0.0049
S(t)	26	1636.7524	30.8957	1584.6097	0.0000

We monitored nests daily from distances of >10 m and we determined fate of the nest as abandoned, successful, or unsuccessful. We defined a depredated nest as any nest in which ≥1 eggs was destroyed and the adult bird did not return to incubate the remaining clutch. A nest was deemed abandoned when the hen did not complete incubation and all eggs were still intact. We defined a nest successful when ≥1 egg hatched.

Statistical Analysis

Survival And Cause-specific Mortality - We used the known-fate model in program MARK (version 5.2; White and Burnham 1999) to explain variation in survival, estimate daily survival rates and estimate the probability of surviving explicit time-periods (e.g., season, year) for male and female bobwhites. The known-fate model employs a binomial likelihood (weekly in our case) and permits incorporation of individual covariates (e.g., gender) delineated by groups (e.g., years in our case) to evaluate their affect on survival. When the fate (alive, dead or censored) of every radio-marked animal is known for each survival interval, the known fates model generates Kaplan-Meier survival estimates (Kaplan and Meier 1958, Pollock et al. 1989). However, because we had missing data for some intervals (e.g., when radio-contact was lost or bobwhites were not checked during a given interval due to stochastic events [i.e., inclement weather]), the variance components of the survival estimates generated from the known-fate model in program MARK are more suitable than those calculated by traditional Kaplan-Meier methods.

We used an information-theoretic approach (Burnham and Anderson 2002, Anderson et al. 2000) to evaluate the set of candidate models. The models were developed *a priori* based on biological insight to avoid superfluous model building (i.e. data dredging). The best approximating model in the set of candidate models was determined by Akaike's Information Criteria (AIC); adjusted for small sample bias and over-dispersion ($QAIC_c$; Burnham and Anderson 2002). We used the median c-hat method as implemented in Program MARK to assess and correct for over-dispersion (c-hat = 1.78) among our data. $QAIC_c$ is a valid model selection method for both nested and non-nested sets of models (Burnham and Anderson 2002). $QAIC_c$ was used to compare each candidate model, and the model with the lowest $QAIC_c$ value was considered to be the best approximating model given the data.

Nest Survival - We estimated daily survival rate

Table 2: Predicted probability of surviving (mean survival and 95% confidence intervals) during overwinter (OW), breeding (Breed) and annual time-periods derived via Program MARK for northern bobwhites located on Whitehall Plantation in Laurens and Bleckley County, Georgia, 1999 - 2002.

Season	DSR[a]	SE[b]	LCI[c]	UCI[d]	Survival	SE	LCI	UCI
OW 1998 - 1999	0.9757	0.0054	0.9625	0.9843	0.5272	0.0748	0.3806	0.6737
Breed 1999	0.9627	0.0073	0.9454	0.9746	0.3717	0.0713	0.2319	0.5116
ANNUAL	0.9697	0.0044	0.9597	0.9773	0.2024	0.0474	0.1094	0.2953
OW 1999 - 2000	0.9693	0.0056	0.9563	0.9786	0.4448	0.0655	0.3164	0.5732
Breed 2000	0.9605	0.0067	0.9449	0.9717	0.3503	0.0625	0.2277	0.4729
ANNUAL	0.9652	0.0043	0.9556	0.9727	0.1583	0.0365	0.0867	0.2299
OW 2000 - 2001	0.9710	0.0054	0.9582	0.9800	0.4656	0.0667	0.3348	0.5963
Breed 2001	0.9600	0.0072	0.9431	0.9720	0.3463	0.0664	0.2162	0.4764
ANNUAL	0.9662	0.0044	0.9565	0.9739	0.1677	0.0392	0.0908	0.2446
OW 2001 - 2002	0.9690	0.0071	0.9517	0.9803	0.4415	0.0815	0.2817	0.6013

[a] DSR is the interval survival 7-days for this study, [b] SE = standard error , [c] LCI = lower 95% confidence interval, [d] UCI = upper 95% confidence interval

(DSR) for bobwhite nests and evaluated competing models explaining variation in nest survival using a general linear mixed model approach (Dinsmore et al. 2002, Stephens 2003, Rotella et al. 2004). We fit models using PROC NLMIXED in SAS because it provided the framework needed to model our binomially distributed data (nest fate = 0 if failed and 1 if successful) and provided a user defined link option (i.e., logit link) while concurrently considering the affects of habitat (PP [planted pines], LH [linear habitats], and other) and gender covariates and, the random effect of year on nest survival (PROC NLMIXED; Institute 1999). We considered year a random effect because we assumed that year was a random level sample and to avoid confounding fixed effects of other variables of interest (e.g., LH, PP).

We used an information-theoretic approach (Burnham and Anderson 2002, Anderson et al. 2000) to evaluate the set of candidate models. The models were developed *a priori* based on biological insight to avoid superfluous model building (i.e., data dredging). The best approximating model in the set of candidate models was determined by Akaike's Information Criteria (AIC); adjusted for small sample bias (AIC_c; Burnham and Anderson 2002). AIC_c is a valid model selection method for both nested and non-nested sets of models (Burnham and Anderson 2002). AIC_c was used to compare each candidate model, and the model with the lowest AIC_c value was considered to be the best approximating model given the data. The relative plausibility of each model in the set of candidate models was assessed by Akaike weights (w_i, Burnham and Anderson 2002, Anderson et al. 2000), where the best approximating model in the candidate set has the greatest Akaike weight (Burnham and Anderson 2002, pg. 447). We used model averaging (Burnham and Anderson 2002, pg. 448) to calculate model averaged coefficients (LH, gender); and we report these coefficients, their standard errors and 95% confidence intervals, and odds ratios.

Habitat Use and Selection - We examined habitat use for bobwhite coveys (1 Oct - 31 Mar) and broods (breeding season) with 2nd and 3rd order habitat selection (Johnson 1980) for individual cov-

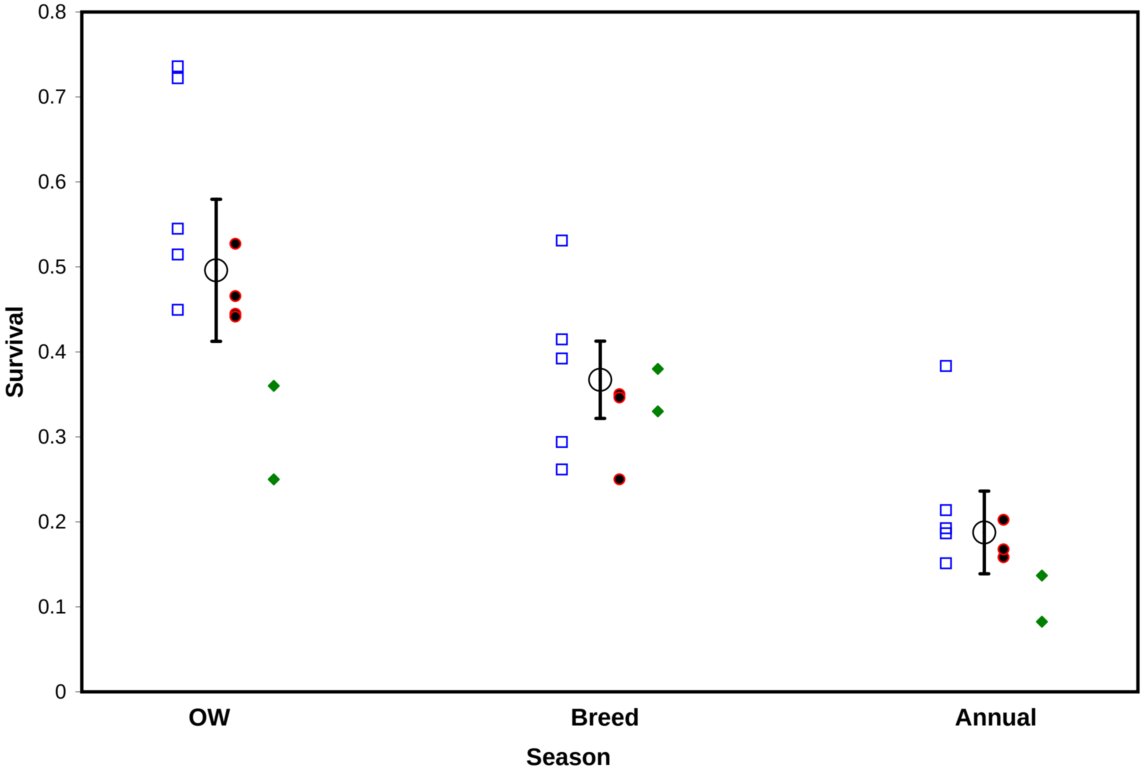

Figure 1: Predicted probability of surviving (mean survival and 95% confidence intervals) during overwinter (OW), breeding (Breed) and annual time-periods obtained via Program MARK (black circles with red outline data points), for our data compared to long-term plantation survival estimates (hollow square-shaped points) and an unmanaged agriculture site (green diamond-shaped points) in southwestern Georgia derived via Kaplan-Meier during 1998 - 2002.

eys and broods, respectively, using compositional analysis (CA; Aebischer et al. 1993, Manley et al. 2000). We defined second order availability for individual coveys and broods (only when $n > 3$ radio-tagged bobwhites/covey). The average habitat proportions within these polygons was calculated and considered to be second order availability. Second-order use was defined as the proportions of each habitat type within home ranges. We defined 3rd order availability as the proportion of each habitat type within home ranges and habitat use as the proportion of individual radio-locations within each habitat type. Prior to analysis, we replaced zero values for use with the value 0.001-an order of magnitude less than the smallest nonzero value (Aebischer et al. 1993). When a habitat was not available for use, we replaced missing values in each log-ratio with the mean of all non-missing values for the respective log-ratio (Aebischer et al. 1993). All habitat selection analyses were conducted using Compos Analysis (version 6.2; Smith 2005). We used a multivariate analysis of variance (MANOVA) test to examine habitat selection (Aebischer et al. 1993). Habitats were ranked using a matrix that indicated the difference of log ratios between habitat types, and log ratio differences were determined with paired t-tests (Aebischer et al. 1993).

We used GIS to assess metrics of habitat composition and configuration using the Animal Movements Extension (AME; Hooge and Eichenlaub 1997) to calculate fixed kernel winter home ranges (Worton 1989) using a 95% isopleth. Kenward (2001, pg. 231)

Table 3: Cause-specific mortality for known-fate radio-tagged northern bobwhites (n = 253) on Whitehall Plantation in Laurens and Bleckley counties, Georgia, 1999 - 2002.

	Causes of Mortality				
	Mammal	Avian	Snake	Harvest	Total
1998-1999	14	47	1	2	64
1999-2000	12	67	0	2	81
2000-2001	13	63	3	2	81
2001-2002	3	22	0	2	27
Pooled	42	199	4	8	253
Percent	16.6	78.66	1.58	3.16	100

indicated that ~20 locations was needed for home range size stability when using the kernel method; thus, coveys and broods with ≤20 locations were excluded from analysis. We also excluded mortality locations from analysis since predators may have transported birds away from the original kill site.

Results

Survival

We monitored 498 bobwhites (n_{female} = 279, n_{male} = 219) during the 3.5-year study. The most parsimonious known-fates model for our data included time-dependency as a constant-seasonal effect (Table 1) with a model weight of 0.47. The model averaged coefficient for breeding season (1 May - 30 Sep) was -0.303 (SE = 0.151). This indicated that breeding season had a negative effect (i.e., survival was lower than over-winter season) on survival for our data. Annual variation in survival was not evident (w = 0.0096; Table 1) for our data. The additive effect of gender to the best model did warrant some consideration (w = 0.2070; Table 1). Whereas the model averaged coefficient estimate for gender (i.e., female) effect was 0.124 (SE = 0.175) indicating that females survived better than males, but the confidence limits for the effect of gender included 0. Further, the model including a season and gender interaction had relatively little support (ΔQAIC$_c$ = 3.09, w = 0.1002) indicating that variation in survival relative to gender was not dependent on season (i.e., breeding or over-winter) for our data.

Generally, over-winter (OW) seasonal survival was higher than breeding (Breed) season survival (Table 2). Breeding season had a negative effect on survival (β = -0.318; 95% CI: -0.614, -0.023). The average OW and Breed season survival was 0.4698 (SE = 0.0721) and 0.3561 (SE = 0.0667), respectively. OW survival was lower than long-term bobwhite estimates from plantations but higher than unmanaged agriculture sites in southwestern Georgia (Figure 1). Breeding season and annual survival was similar among sites (Figure 1). Avian species were the most prevalent agent of mortality accounting for 78.66% of the known-fate bobwhites (Table 3). Mammals accounted for 16.60% of bobwhite mortalities, whereas snakes and harvest combined for <5%.

Reproduction

We monitored 165 nests (n_{1999} = 45, n_{2000} = 72, n_{2001} = 48) during the 3.5-year study. The constant-among years-model was the best supported model for our data (Table 4); however, models including individual-additive fixed effects of gender, PPs, LHs and a random year-effects model warranted consideration (ΔAIC$_c$ ≤2, w > 0.1000). The model

Table 4: Mixed models (i.e., fixed and random effects models) explaining nest survival treating year as fixed and random effects; and, linear habitats (LH) and gender as fixed effects for northern bobwhites located on Whitehall Plantation in Laurens and Bleckley County, Georgia, 1999 - 2002.

Model	K	AIC	AIC_c	$\Delta\, QAIC_c$	W_i
B_0	1	604.8676	604.8695	0.0000	0.3013
$B_0 + B_1{}^*(PP)$	2	606.7752	606.7808	1.9113	0.1159
$B_0 + B_1{}^*(Gender)$	2	606.8171	606.8226	1.9531	0.1135
$B_0 + B_1{}^*(LH)$	2	606.8453	606.8508	1.9813	0.1119
$B_0 + u$	2	606.8676	606.8732	2.0037	0.1106
$B_0 + B_1{}^*(Year1) + B_2{}^*(Year2)$	3	607.9712	607.9823	3.1128	0.0635
$B_0 + B_1{}^*(PP) + B_2{}^*(LH)$	3	608.7739	608.7849	3.9154	0.0425
$B_0 + u + B_1{}^*(PP)$	3	608.7752	608.7863	3.9168	0.0425
$B_0 + u + B_1{}^*(Gender)$	3	608.8171	608.8281	3.9587	0.0416
$B_0 + u + B_1{}^*(LH)$	3	608.8453	608.8563	3.9869	0.0410
$B_0 + u + B_1{}^*(PP) + B_2^*(LH)$	4	610.7739	610.7923	5.9228	0.0156

considering year as a fixed effect had relatively little support (ΔAIC$_c$ = 3.11, w =0.0731) compared to other top-ranked models. Therefore, we primarily fit models treating years as random effects so as not to confound with other fixed effect parameters.

We used model averaging to interpret coefficients for nest survival and individual covariates (Table 5). The odds ratios for gender, PP and LH were 1.07, 1.07 and 0.95 (Table 5), respectively, indicating that females and nests located in PPs were 7% more likely to be successful than nests incubated by males and found in other habitats, respectively; however, the confidence interval for these log ratios included 1. Daily nest survival (DSR) for 1999, 2000, and 2001 was 0.9727 (SE = 0.0065), 0.9645 (SE = 0.0055), and 0.9705 (SE = 0.0065), respectively. The average DSR for years pooled was 0.9687 (SE = 0.0037). Nest survival for years pooled delineated by gender and habitat type was higher for females and nests located in PPs, although these differences were not significant (Figure 2).

Habitat Use

Coveys - We combined all coveys (n = 67) during the 3.5-year study for habitat selection analysis; we determined that habitat selection did not differ between years ($F_{2,66}$ = 1.58, P = 0.214). Covey habitat selection departed from random at both the second-order (λ = 0.6467, χ^2_3 = 29.206, $P < 0.001$) and third-order (λ = 0.2644, χ^2_3 = 89.139, $P < 0.001$) levels. For our data, coveys preferred PP types over hardwoods and miscellaneous types (Table 6; 2nd order: t_{66} = 3.515, $P < 0.001$; 3rd order: t_{66} = 5.870, $P < 0.001$) and AG/FAL habitat types (Table 6; 2nd order: t_{66} = 3.628, $P < 0.001$; 3rd order: t_{66} = 9.580, $P < 0.001$). LH habitat types was preferred to hardwoods and miscellaneous habitats, but the difference was not significant at the 3rd order level (Table 6; 2nd order: t_{66} = 2.921, P = 0.005; 3rd order: t_{66} = 0.220, P = 0.827), and LH was preferred to AG/FAL habitat (Table 6; 2nd order: t_{66} = 4.247, $P < 0.001$; 3rd order: t_{66} = 3.181, P = 0.002). In order of preference at the second order level bobwhite coveys preferred: planted pines, linear habitats, managed woodlands, hardwoods and thinned hardwoods, agricultural and fallow land. And at the

Table 5: Model averaged, estimated coefficients and associated precision for parameters used to model variation in nest survival for northern bobwhites located on Whitehall Plantation in Laurens and Bleckley County, Georgia, 1999 - 2002.

			Confidence Interval		
Parameter	Estimate	SE	Lower 95%	Upper 95%	Odds Ratio
Gender (female)	0.0710	0.3136	-0.5437	0.6856	1.0736
Planted Pine (PP)	0.0764	0.2518	-0.1755	0.3282	1.0793
Linear Habitat (LH)	-0.0486	0.3234	-0.3720	0.2748	0.9525

third order level bobwhite coveys preferred: planted pines, managed woodlands, linear habitats, hardwoods and thinned hardwoods, agricultural and fallow land.

Broods - We combined all broods ($n = 73$) to examine habitat selection and preference for the 3.5 year study. Brood habitat selection was not random at both the second-order ($\lambda = 0.2631$, $\chi^2_3 = 97.470$, $P < 0.001$) and third-order ($\lambda = 0.2632$, $\chi^2_3 = 97.441$, $P < 0.001$) levels. Broods preferred LHs over all other habitat types at the second-order level and all other habitat types except PPs at the third-order levels (Table 7). At the second-order level: LHs were preferred to agriculture and fallow habitats although the difference was not significant ($t_{72} = 1.034$, $P = 0.302$), PPs ($t_{72} = 3.051$, $P = 0.003$), hardwoods and other habitats ($t_{72} = 12.906$, $P < 0.001$), and managed woodlands ($t_{72} = 2.867$, $P = 0.005$); PPs was preferred to hardwoods and other habitats ($t_{72} = 7.859$, $P < 0.001$), and managed woodlands ($t_{72} = 2.867$, $P = 0.005$); agriculture and fallow lands was preferred over hardwoods ($t_{72} = 10.132$, $P < 0.001$), managed woodlands ($t_{72} = 2.034$, $P = 0.046$), and planted pines (PPs) although the difference was not significant ($t_{72} = 1.785$, $P = 0.085$); and managed woods was preferred over hardwoods ($t_{72} = 6.583$, $P < 0.001$). At the third-order level: PPs was preferred to agriculture/fallow land ($t_{44} = 4.672$, $P < 0.001$), hardwoods and other habitats ($t_{10} = 7.709$, $P < 0.001$), managed woodlands ($t_{34} = 2.876$, $P = 0.007$), and LHs, although the difference was not significant ($t_{46} = 0.371$, $P = 0.713$); and LHs was preferred to agriculture habitats ($t_{54} = 4.688$, $P < 0.001$), hardwoods and other late-succession habitats ($t_{12} = 2.154$, $P < 0.050$), and managed woodlands although the difference was not significant ($t_{72} = 1.303$, $P = 0.200$).

Population Response

We used covey call counts via the point-count method (Wellendorf et al. 2004) to determine bobwhite abundance. Using a replicated design and 14 individual, fixed points we estimated the initial bobwhite abundance at 0.86 birds/ha (~45 coveys). During fall 2001, we estimated a final bobwhite abundance of 1.48 birds/ha. Thus, we observed an estimated 75% increase in bobwhite abundance during the 3.5-year study.

Discussion

Survival

Bobwhite survival has been documented to vary both temporally and spatially (Burger et al. 1995*a*, 1998, Curtis et al. 1988, Sisson et al. 2009, Taylor et al. 2000, Terhune et al. 2007) and relative to gender (Pollock et al. 1989). During this study, variation in survival was best explained by models including seasonal effects. There was a negative effect of breeding season on bobwhite survival; bobwhites were 1.37 times (37%) less likely to survive during breeding

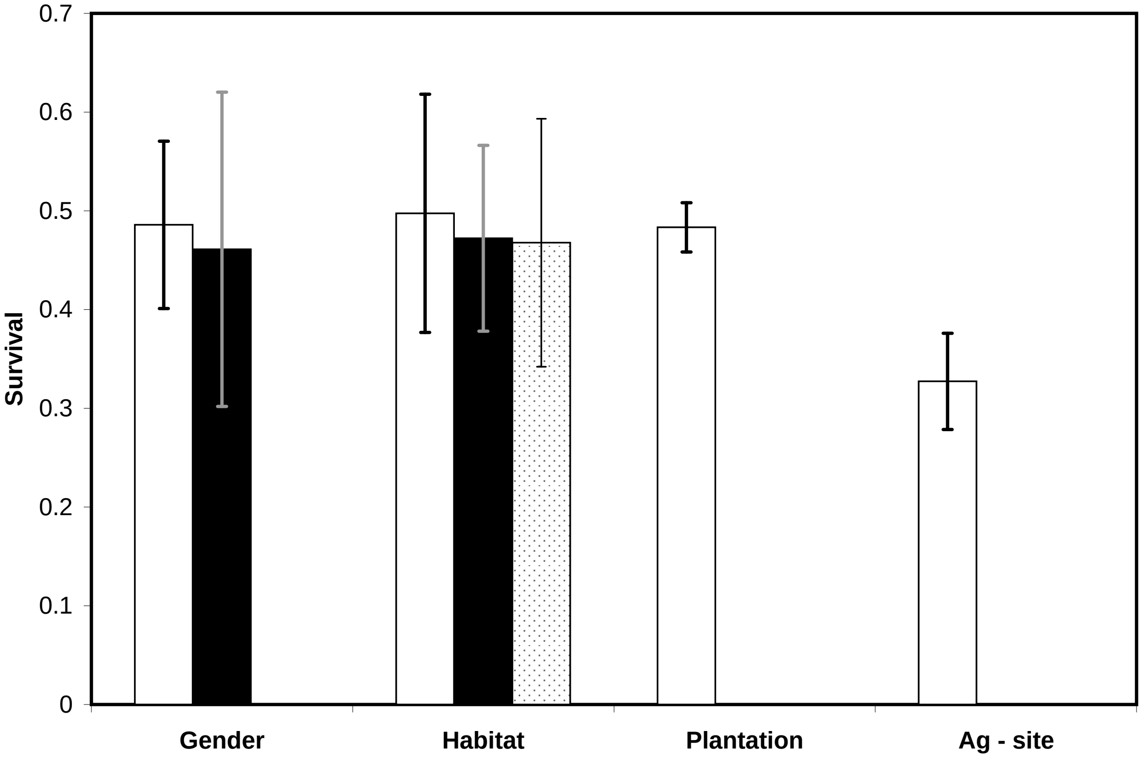

Figure 2: Predicted probability of survival (mean survival and 95% confidence intervals) of northern bobwhite nests as estimated via model averaging for gender (male [solid black bars], female [solid white bars]) and habitat (LH [black-speckled bars], other habitats [solid black bars], and PPs [solid white bars]) on Whitehall Plantation in Laurens and Bleckley County, Georgia, compared to nest survival estimates for an intensively managed plantation and an unmanaged agriculture site during 1999 - 2002.

season than OW season. Whereas the most parsimonious model including gender did warrant some consideration, the model including the interaction of gender and season was not adequately supported (Δ QAIC$_c$ = 3.09). This suggested that survival relative to gender was not dependent on season. Our survival estimates were similar to those of other reported studies (Curtis et al. 1988, Burger et al. 1998, Sisson et al. 2009, Terhune et al. 2007). Breeding season and annual survival during our study was, in general, similar to intensively managed plantation sites and an unmanaged agriculture site (Figure 1); however, OW survival was markedly disparate between sites. Interestingly, survival during OW declined relative to management strategy whereby intensively managed plantation sites, managed agriculture sites (i.e. our study site), and unmanaged agriculture sites incurred the highest, near average, and lowest survival, respectively. Similarly, Cook (2004) demonstrated that bobwhites exhibited higher survival on areas with linear habitats compared to those void of them. As such, the utility for augmenting habitat among agricultural landscapes via novel habitats (i.e. PPs and LHs) to improve survival, at least compared to unmanaged agricultural landscapes, was substantiated by our critique.

Fies et al. (2002) suggested bobwhite dispersal and movement proclivities are potentially greater among fragmented landscapes and Kabat and Thompson (1963) reported increased movements of bobwhites when landscapes typically consisted of marginal habitat. Incidentally, Cook (2004) reported that bobwhites on sites with linear habitats exhibited lower dispersal rates than sites without these

Table 6: Simplified ranking matrices for northern bobwhite coveys based on second- and third-order compositional analysis on Whitehall Plantation located in Laurens and Bleckley counties, Georgia 1999-2002.

	Ag / Fall[a]	PP[b]	LH[c]	HW / Other[d]	MW[e]	Rank
2^{nd} Order habitat selection (n = 67)[f]						
Ag / Fall		—	—	-	—	0
PP	+++		+	+++	+	4
LH	+++	-	+	+++		3
HW / Other	+	—	—		—	1
MW	+++	-		+++	-	2
3^{rd} Order habitat selection (n = 67)[g]						
Ag / Fall		—	—	-	—	0
PP	+++		+++	+++	+	4
LH	+	—	+		—	2
HW / Other	+++	—		-	—	1
MW	+++	-	+++	+++		3

[a] Ag / Fall denotes habitat types including agriculture and fallow fields, [b] PP represents planted pines (typically longleaf), [c] LH represents linear habitats: field buffers and borders, hedgerows, linear longleaf pines, and terraces, [d] HW denotes habitat types including hardwoods and other habitats not typically associated with early succession vegetation (e.g., drains), [e] MW represents managed woods: upland pines, early succession vegetation areas other than fallow fields, burned and unburned habitats, and thinned and managed mixed hardwood pine stand, [f] 2nd order analysis was based on comparing the proportional habitat use within home ranges with the proportion of total available habitat types (i.e., study area vs. home range); a triple sign indicates a significant deviation from random at the alpha level of 0.05 and positive and negative signs indicates habitat preference and avoidance, respectively, [g] 3rd order analysis was based on comparing the proportional habitat use within home ranges with the proportion of telemetry locations located within each habitat type within each bird's home range (i.e., home range vs. locations).

habitats. Additionally, Sisson et al. (2000, 2002) demonstrated that survival and home range size was dependent on resource quality and availability. Hughes et al. (2005) reported evidence to support these notions: they suggested that due to lack of resources (e.g. habitat and food availability) bobwhites were forced to utilize lower quality habitats (e.g. creek swamps, hardwoods) and traverse unsuitable habitat(s) to get to suitable habitat and/or food; however during years of abundant food resources they determined that home range size and survival improved dramatically. Although not reported herein, bobwhites on our study site generally retained high site fidelity; home range size was only marginally larger than those on intensively managed plantation sites and smaller than those on unmanaged agriculture sites (S. Mitchell, Alabama Quail Project, unpublished report), indicating that resource availability was likely not a limiting factor during our study - although supplemental feeding did occur on our study site and thus home range size may have been low from this highly available resource (Sisson et al. 2000). Collectively, novel habitats (e.g. PPs and LHs) may decrease home range size, improve survival, and reduce dispersal rates among fragmented and/or agricultural landscapes.

Covey Habitat Use - Bobwhite coveys preferred PP habitats, managed woodlands, and LHs to all other available habitats (Table 6). These findings were not contrary to what we expected because during OW months (1 Oct - 31 Mar) a significant portion of the agriculture area was disked under and was thus bare soil. The preference of PP and LH (3rd Order selection) habitats over managed woodlands

Table 7: Simplified ranking matrices for northern bobwhite broods based on second- and third-order compositional analysis on Whitehall Plantation located in Laurens and Bleckley counties, Georgia 1999-2002.

	Ag / Fall[a]	PP[b]	LH[c]	HW / Other[d]	MW[e]	Rank
2nd Order habitat selection (n = 73)[f]						
Ag / Fall		+	-	+++	+++	3
PP	-		—	+++	+	2
LH	+	+++		+++	+++	4
HW / Other	—	—	—		—	0
MW	—	-	—	+++		1
3rd Order habitat selection (n = 73)[g]						
Ag / Fall		—	—	+	-	2
PP	+++		+	+++	+++	4
LH	+++	-		+	+	3
HW / Other	-	—	-		+	1
MW	+	—	-	-		1

[a] Ag / Fall denotes habitat types including agriculture and fallow fields, [b] PP represents planted pines (typically longleaf), [c] LH represents linear habitats: field buffers and borders, hedgerows, linear longleaf pines, and terraces, [d] HW denotes habitat types including hardwoods and other habitats not typically associated with early succession vegetation (e.g., drains), [e] MW represents managed woods: upland pines, early succession vegetation areas other than fallow fields, burned and unburned habitats, and thinned and managed mixed hardwood pine stand., [f] 2nd order analysis was based on comparing the proportional habitat use within home ranges with the proportion of total available habitat types (i.e., study area vs. home range); a triple sign indicates a significant deviation from random at the alpha level of 0.05 and positive and negative signs indicates habitat preference and avoidance, respectively, [g] 3rd order analysis was based on comparing the proportional habitat use within home ranges with the proportion of telemetry locations located within each habitat type within each bird's home range (i.e., home range vs. locations).

was likely a result of the timing of the study. The managed woodlands were heavily disturbed when they were logged and cleaned up therefore producing mostly weeds during the first couple of years. This made good summer habitat but had not yet developed into good winter cover. We speculated that bobwhites utilized PP habitats at a higher than expected rate because of the woody vegetation component provided via the longleaf pines and the fact that groundcover was more fully developed, thereby improving the quality of "escape" cover for coveys.

Among agriculture landscapes a paucity of suitable bobwhite habitat exists throughout the year. This was evident by the extremely low OW survival and large home range sizes observed on agricultural sites without PPs and LHs or newly created early-succession habitats (Hughes et al. 2005) when compared to OW survival for our site where these habitats were available during the entire study. Furthermore, breeding season survival did not vary among sites, irrespective of management strategy, indicating that PP habitat was more critical during OW seasons, a time when habitat is likely a limiting factor among agricultural landscapes. Because bobwhites are considered an r-selected species (demonstrated by high annual mortality and high reproductive output), OW survival has been recognized as a vital demographic parameter for increasing bobwhite populations (Burger et al. 1998, Sisson et al. 2009). Under this tenet, by increasing OW survival, whether via habitat management or other means, the number of bobwhites available to reproduce is potentially augmented; and thus, improving reproductive output and subsequently increasing bobwhite abundance.

Therefore, the utility of creating novel habitats (PPs and LHs) and improving existing habitat as demonstrated in this study, under this tenet alone, may immensely improve bobwhite abundance among agricultural landscapes and other OW-habitat deficient sites.

Cause-specific Mortality - Despite numerous studies reporting agents responsible for mortalities of bobwhites, cause-specific mortality remains an enigma and is one of high observer subjectivity. Therefore, in this study, we only report assessed causes of mortality and compare our results to those studies conducted by the AQP (where protocols for ascribing causes of mortality were similar) to limit observer variability.

During this study, avian depredation was the leading cause of mortality for all years accounting for nearly 80% of all known-fate mortalities. Mammals accounted for approximately 17% and snake and harvest combined for <5%. Our results were generally similar to those reported by Sisson et al. (2009); however, avian mortality was elevated for our study site compared to their long-term results. Surprisingly, differences in causes of mortality did not vary relative to season for our data; avian species remained the leading mortality agent during both breeding and OW season. Notably, other sites exhibited variation in causes of mortality relative to season, whereas during breeding season mammals typically became a more salient cause of mortality than during OW seasons (Sisson et al. 2009). For example, Sisson et al. (2009) reported that one site in east-central Georgia experienced high OW-avian mortality (>71%) and high breeding-season mortality caused by mammals (>61%). Particularly noteworthy was the unmanaged agriculture site: avian mortality during breeding season (>61%) was much higher than mammalian mortality (<34%). Perhaps avian mortality, while ostensibly dependent on the timing and duration of raptor migration, is higher on agriculture sites whether or not PP and LH habitats are present. In such cases, habitat composition and juxtaposition may play a key role in the efficiency of avian predators to locate (and depredate) bobwhites among these types of landscapes. Thus, more research is warranted to determine whether wider linear habitats may mitigate avian mortalities and/or whether other proximate habitats (e.g. hardwoods) decrease the utility of novel habitats on specific sites.

Reproduction

Daily survival rates for northern bobwhite nests in our study did not vary among years for our data (Table 4). Since we were interested in nest survival among PPs and LHs compared to other habitat types, we treated year as a random effect to evaluate habitat type and gender effects on nest survival. The most parsimonious model was a constant survival model with no covariate effects. Our nest survival estimates were higher than those reported for other nest studies (Burger et al. 1995*b*, Puckett et al. 1995, Hughes et al. 2005), and similar to long-term nest survival estimates for intensively managed plantations (Figure 2). However, mammalian nest predator management did occur on these study sites.

Hughes et al. (2005) surmised that lack of resource availability, particularly habitat availability, was a limiting factor during their study, and suggested that habitat development - such as field borders, field buffers, and hedgerows as well as other habitat practices (e.g., no-till farming) - would benefit bobwhite nest survival and production. Previous researchers have indicated that nest predators may more efficiently forage in landscapes comprised of small and/or narrow habitats (Puckett et al. 1995, 2000). Additionally, Puckett et al. (1995) reported low nest success for nests located in filter strips, particularly during the early nesting season. We did not, however, find evidence suggesting that linear habitats negatively impacted nest survival (Figure 2). Notably, when compared to our study, the effective land area and width of filter strips was different for the study conducted by Puckett et al. (1995, 2000), mean filter strip width was 9.2 m and the effective land area comprised of filter strips was <10 percent. Further, filter strips were designed to reduce soil erosion and thus oftentimes were located along ditches (Puckett et al. 1995) - a habitat conducive to certain predators (e.g., snakes). In contrast, during our study, the effective land area created from PP (21%) and LH (12%) habitat development was >30% and the mean LH width was 15 m. Therefore, implementation of wider LHs and increased effective land area may improve nest survival, increasing the amount of habitat for nest predators to rummage. As such, recent implementation has demonstrated that when the effective land area was increased and wider LHs were constructed, bobwhite demographics and population levels increased among agricultural landscapes (D. C. Sisson, Albany Quail Project, unpublished report).

Among PP habitats, longleaf pines provided woody substrate and pine needles for nest building; nearly 42% of all nests during this 3.5-year study were located in PP habitats (S. Mitchell, Alabama Quail Project, personal communication) and nest survival among these habitat types was highest during our study. When combined with the nests located in LHs nearly 64% of all nests were constructed and incubated in these newly created habitat types. Thus, the development of these habitat types minimally improved the quantity of habitat available during nest season and, seemingly, did not render bobwhite nests more susceptible to predation.

Brood Habitat Use - Bobwhite hens preferred to raise broods in LHs, PPs and fallow areas compared to other habitat types, and they used PPs and LHs more than agriculture sites (Table 7). The higher use of LHs and PPs compared to agriculture cropland may have been attributed to later cover availability via crops in those areas combined with pesticide use (and low arthropod availability). Our results, for brood habitat use, were similar to those reported for other studies (Puckett et al. 1995, 2000, Cook 2004).

Cook (2004) and Puckett et al. (2000) reported that hens raising broods used LHs more than agriculture fields and other habitat types. Puckett et al. (2000) also reported that bobwhite chick survival was high among LHs and brood home range sizes were small. Although not reported herein, we observed that, in general, bobwhite brood home-range size was similar to those of intensively managed plantation broods (S. Mitchell, Alabama Quail Project, personal communication). Therefore, we surmised that given the preference for LHs and PPs, and similar home range size of broods during our study when compared to other intensively managed sites that these habitat types may facilitate reduced home range size for broods in agriculture landscapes. Thus, provided herbicides and pesticides are not exploited in these habitats, LHs and PPs may render the much needed niche for bobwhite broods in agriculture ecosystems whereby weedy vegetation and arthropods are prevalent and year-round habitat is made available.

Summary

In this study, we reported data that advocated novel habitat (e.g. PPs and LHs) establishment as a practical utility to promote improved demographics when compared to intensively managed plantation sites and unmanaged agriculture sites: survival was generally similar to managed sites and higher than unmanaged sites; reproduction was similar to managed sites and higher than unmanaged sites; habitat use by broods and coveys was high among novel habitats and, while anecdotal, broods and coveys benefited from PPs and LHs by reducing their home range size and providing the much needed resource availability during germane times (i.e. OW months). And, these novel habitats effectively rendered nesting habitat during breeding season and nest survival was similar among these habitat types compared to other habitat types. The combined effect of novel habitat establishment and improvement of existing habitat provided a substantial overhaul to the landscape on this study site. We effectively managed >60% of the landscape to benefit bobwhites, while maintaining farm practices on much of the remainder. As a result, bobwhites utilized novel and improved habitats heavily for covey home ranges, nest sites, and brood habitat which resulted in increased bobwhite abundance and a renewed optimism for managing bobwhites outside the traditional "plantation belt" located in southwest Georgia and north Florida. Bobwhite abundance increased each year following habitat modification whereby point counts conducted during the fall (Oct-Nov; Wellendorf et al. 2004) indicated an increase by >75 percent. Bobwhite abundance increased from <0.86 birds per hectare to >1.48 birds per hectare during the 3.5-year study.

Management Implications

Researchers and biologists have demonstrated that the most effective mode to restore bobwhite populations, both at local and regional scales, is to increase habitat availability (Klimstra 1972, Brennan 1991). Likewise, long-term research corroborates this notion where, despite the declining status of bobwhites throughout most of their range during the past decade, intensively managed bobwhite plantations have experienced stable-to-increasing bobwhite abundance (Brennan 1991, Stribling and Sisson 2009). The results from this study also illustrated the importance of habitat management to benefit bobwhites. Thus, for bobwhites to persist among agricultural landscapes, restoration (i.e., Farm Bill - CRP) programs should continue to focus on habitat management. Whereas implementation of novel habitats is by no means a panacea for reversing population declines, they may serve as pragmatic utility for at least improving bobwhite habitat among agricultural landscapes and perhaps extenuate bobwhite population declines among these ecosystems. Further, several other techniques (i.e. conservative bobwhite harvest, nest predator management, supplemental feeding) when used in conjunction with establishing novel habitats among agriculture ecosystems, may also increase restoration success.

When establishing linear habitats among agricultural landscapes, we recommend setting the target of land area affected at a minimum of 10-15% and linear habitat widths $\geq$15 m (and when applicable wider). We also recommend employing other habitat management techniques in conjunction with linear habitat establishment when appropriate: managing dry corners for early-succession vegetation, timber density reduction on adjacent sites, mid- and over-story hardwood reduction, prescribed burning, supplemental feeding and nest predator management. When planting pines, we recommend planting longleaf pines at a conservative (8X8 or greater; 600 trees/acre or less) spacing, and utilizing prescribed fire and limb pruning as needed to benefit early-succession vegetation over time. Proper management of woodlands surrounding agricultural areas should also greatly improve habitat conditions among agricultural ecosystems - this type of management was a large part of the success observed during our study. Additionally, we encourage federal and state programs implementing early-succession habitat establishment to concentrate efforts to specific-focal areas, particularly areas located near existing bobwhite populations, and expand outward in order to maximize restoration efforts at both the local and regional scale. Lastly, we recommend continued research and monitoring of bobwhite populations among these types of ecosystems to continue gleaning insight about the utility of novel habitats and learn how these ecosystems function, as a whole, both locally and regionally.

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Habitat Management

Half-Cutting as a Management Tool to Increase Abundance of Northern Bobwhite in South Texas

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We evaluated the effectiveness of half-cutting honey mesquite (*Prosopis glandulosa*) trees to increase northern bobwhite (*Colinus virginianus*; hereafter bobwhite) habitat and abundance in South Texas. We compared the effects of half-cutting on the survival of mesquite and its effects on understory vegetation on both treated and control areas monthly. Data were taken under the tree's canopy to determine understory plant species diversity, height, and density. We used bobwhite whistle counts, mark-recapture, and searched with trained dogs to determine the effects of half-cutting on bobwhite abundance. Half-cut young trees had 23.1% greater survival than did older half-cut trees. The area protected from grazing under half-cut trees was 10.2 times larger than that protected by control trees. Height of understory vegetation under half-cut trees was significantly (P = 0.005) taller then that under control trees. Treated areas supported more plant species than control areas but the difference was not significant (P = 0.072). Three bobwhite food plants had a positive response to half-cutting, whereas 3 were negatively affected. Half-cutting had no significant influence on numbers of cocks whistling (maximum 9 and 8 males calling/5-min period, respectively, for treatment and control). The number of bobwhite trapped on the half-cut area was 91 while only 75 were trapped on the control area. Trained bird dogs located 101 bobwhites on the half-cut areas and 78 on the control areas. The half-cut areas had the same number of coveys (10) as the control areas, however, covey sizes were larger (2.3 birds/covey) on the half-cut areas. We believe that half-cutting can add habitat and increase bobwhite abundance on many heavily grazed rangelands.

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Key words: abundance, *Colinus virginianus*, habitat, half-cutting, management, northern bobwhite, Texas

Introduction

It is common practice in the United States to manipulate habitat in an attempt to increase northern bobwhite population numbers. Ellis (1972) obtained responses from 25 states on manipulation techniques applied in managing bobwhite habitat and noted the most popular methods included (popular to least most): herbaceous planting, shrub planting, burning, timber clearing, disking, controlling brush, and prescribed grazing. Rosene (1969) described most of these methods and how they were applied to landscapes in the southeastern United States. Jackson (1969) and Webb and Guthery (1982) conducted studies evaluating the effects of several habitat manipulation techniques on quail in the west Texas Rolling Plains.

Reid et al. (1977) concluded that mesquite offered important habitat for bobwhite nesting, but Jackson (1969, p. 27) noted that mesquite alone did not have much value as quail cover. Rather, the value of mesquite was determined by the degree to which it repelled livestock from the understory. Webb and Guthery (1982) noted a 2.2-fold increase in use by quail of an area due to several manipulation techniques, including half-cutting. However, half-cutting was not studied independently of other habitat manipulation techniques. The objective of this study was to monitor and document the response of vegetative understory in half-cut areas and bobwhite abundance to the practice of half-cutting honey mesquite trees in South Texas.

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Methods

Study Area

The study area was located on the Blocker Energy Corporation's San Tomas Hunting Lease (about 9,500 ha) on the Encino Division of the King Ranch, Brooks County, Texas, USA. This area has been described as the South Texas Plains (Gould 1975), the Rio Grande Plain (Kiel 1976), and it is known locally as the South Texas Brush Country. In 1981, the San Tomas Hunting Lease provided 442 hunter days for 200 hunters. From 1979-1983, quail densities on the lease ranged form 2.5 to 5.1 birds/ha as determined with yearly line-transect surveys (Hall 1983). Quail harvests varied from 2,000 to 6,000 quail per season.

Climate, for the majority of the year is maritime turning continental in winter (Beasom 1973). The growing season is about 300 days (December through February being considered winter) with a 31-year average of 57.4 cm of precipitation (King Ranch, Inc., unpublished records). Peaks of rainfall occur in the spring and fall when tropical storms and hurricanes are active, inundating low areas of the level to gently sloping topography at approximately 10-year intervals. The elevation of the study area is between 15-18 m above sea level.

Soils were similar throughout the study area, were sandy and well-drained, and of the Nueces-Sarita Association (Soil Conservation Service 1973). Mesquite trees were of equal age structure and density on both areas and had similar vegetative composition. Sites were dominated by little bluestem (*Schizachyrium scoparium*), paspalum (*Paspalum setaceum*), Pan American balsomscale (*Elyonuris tripsacoides*), lovegrasses (*Eragrotis* spp.) and roundseed dicanthelium (*Dicanthelium sphaerocarponn*). These sites received a year-round grazing pressure of approximately 1 animal unit/6.5 ha.

We conducted our study in a pasture that, prior to 1969, was dominated by old-growth honey mesquite. In 1969, the pasture was root-plowed in an east-west direction. Strips of mesquite 47-140 m wide every 0.8-1.6 km were left to prevent wind erosion, provide shade for livestock, and furnish cover for wildlife (Lehmann 1960).

Half-cutting

We used half-cutting techniques as described by Jackson (1969) and cut trees before foliation from 2-5 March 1982. We cut all limbs >10 cm diameter with a machete on all trees within 100 m of the tree line center. We cut all vertical limbs below 76 cm and all horizontal limbs below the cutter's shoulders. Limbs were cut and bent over in a manner that produced a wagon-wheel effect with all limbs touching the ground. Because mesquite trees have thorns, the bent over limbs protected vegetation growing under them from cattle grazing.

We randomly selected 1-km sections of the "tree lines" as study sites for the half-cutting experiment. We cut limbs and bent them over in a manner that produced a wagon-wheel effect with all limbs touching the ground. We located all treatment tree lines a minimum of 4 km from randomly selected control tree lines. Both treatment and control tree lines were in the same pasture so all other variables such as grazing pressure were held constant.

Understory Vegetation

We randomly selected 5 half-cut trees and 5 multi-stemmed trees suitable for half-cutting for observation on both the treated and control areas each month (9 months total) to compare the effects that half-cutting had on understory vegetation. We used 2 randomly selected 0.1-m^2 quadrats under each tree's canopy to determine understory plant species composition, maximum individual species height (measured in centimeters), and understory species density. We used a paired *t*-test (Ott 1993) to determine if there were significant ($P < 0.05$) differences in understory height or density between treated and control areas. Because of time constraints, we only sampled 200 (66%) of the 305 half-cut trees to document survival of limbs and trees 9 months post-treatment.

Bobwhite Census Techniques

We used 3 methods to obtain indices of bobwhite abundance on the half-cut and control areas. We conducted all surveys within 100 m on each side of the tree lines. We recorded the number of whistling cocks (Rosene 1957) per 5-min period from the last week in April to the end of August in the center of each 1 km tree line (either half-cut or control) during days when the wind was less than 8 km/hour and there was no rain. We monitored each station for 2 consecutive days per week. We began each survey 15 min before sunrise and ended each after 90 min of observation (Wilson and Crawford 1987). We used wire funnel-traps baited with sorghum (milo) to capture bobwhites, and we marked individual bobwhites with an aluminum leg band. We placed traps within the tree lines on both the treatment and control areas in the morning and evening to avoid high mid-day temperatures. We placed all traps under trees to provide overhead protection and shading. We ran traps 15 times from 22 July to 6 October. We surveyed treatment and control areas for bobwhite from 18 October to 13 November using trained dogs to find and hold coveys (Ellis et al. 1969). We counted all individuals in the covey as they flushed.

Results

Half-cutting

Of 305 mesquite trees cut, 152 were rough-barked (older trees) and 153 were smooth-barked (younger trees). On average 15.5 limbs were cut on half-cut trees. Nine months after treatment, 56% of limbs on smooth-barked trees survived as compared to only 35% of limbs on rough-barked trees.

Understory Vegetation

The area protected from grazing under the half-cut trees ($\bar{x}$ = 14.3 m^2, SE = 3.2, n = 45) was 10.2 times larger in area than the area protected by control trees ($\bar{x}$ = 1.4 m^2, SE = 0.3, n = 45). Mean height ($\bar{x}$ = 75 cm, SE = 0.42, n = 45) of understory vegetation under half-cut trees during the 9 months of the study was taller ($P = 0.005$) than mean height ($\bar{x}$ = 61 cm, SE = 0.37, n = 45) of understory vegetation under control trees. During the 9 months of study, understory vegetation consisted of 51 species for both the half-cut and control areas combined. The treated area supported more species ($\bar{x}$ = 26.3, SE =1.34, n = 45) than the control area ($\bar{x}$ = 23.0, SE = 1.39, n = 45), but the difference was not significant (P = 0.072). Both the treated and control areas supported 10 of the most common bobwhite food plants in the area, but they occurred more frequently ($\bar{x}$ difference = 4.7%, SE = 5.10, n = 45) and averaged 8.1 cm (SE = 4.30, n = 45) taller under the protected canopies (Table 1). The major food plants that responded positively were crotons (*Croton* spp.), fringed signalgrass (*Brachiaria ciliatissima*), ragweed (*Ambrosia* spp.), thin paspalum (*Paspalum setaceum*), partridgepeas (*Cassia* spp.), erect dayflower (*Commelian erecta*), and hoary milkpea (*Galactia canescensa*) (Table 1). Food plants that decreased in frequency relative to the control sites were cowpen daisy (*Verbersina* spp), American snoutbean (*Rhynchosia americana*), and tomatillo groundcherry (*Physalis ixocarpa*).

Bobwhite Abundance

Bobwhite whistling-cock surveys indicated June was the peak calling period for all study areas. The number of calling males peaked on 9 June and dropped to 0 on 28 July 1982. Half-cutting had no significant ($P = 1.000$) influence on numbers of cocks whistling (maximum 9 and 8 males calling/5-min period, respectively for treatment [$\bar{x}$ = 5.1, SE = 2.3, n = 12] and control [$\bar{x}$ = 5.1, SE =1.8, n = 12]). During 15 trap-days, the number of bobwhite trapped on the half-cut area was 91 while 75 were trapped on the control area. Thirty-three recaptures also was recorded for both the half-cut and control areas. Bobwhites were counted 3 times on the treatment and the control areas using the trained bird dogs with 101 ($\bar{x}$ = 33.7, SE = 1.3, n = 3) bobwhites located on the half-cut areas and 78 ($\bar{x}$ = 26.0, SE = 2.7, n = 3) located on the control areas. The half-cut areas had the same number of coveys (10) as the control areas, however, covey sizes were larger (2.3 birds/covey) on the half-cut areas.

Discussion

Half-cutting mesquite was a valuable tool for increasing bobwhite habitat on our study area. Although the number of males calling did not support

Table 1: Monthly frequency (%) and height (cm) of major food plants on the half-cut and control areas, Brooks County, Texas, 1982.

Food plants	Mean Frequency Half-cut	Control	Mean Height Half-cut	Control
Croton	86	52	62.2	53.4
Tropic croton	56	34	48	31.7
Cowpen daisy	18	42	95	58
Fringed signalgrass	6	2	13	20
Ragweed	46	24	52.3	49.4
American snoutbean	26	44	13.5	15.9
Thin paspalum	36	32	42.6	20.6
Partridgepea	12	12	32.4	17.8
Erect dayflower	24	18	26.5	15.3
Hoary milkpea	6	2	6	18
Tomatillo groundcherry	32	34	14.2	16.7
Mean	32	27	36.9	28.8

this assumption, trapping and counting quail with dogs indicated that bobwhite used the treated areas 1.2 and 1.3 times more often than the control, respectively. The half-cut areas also had the most calling cocks on a given census (9 on 9 June 1983).

Cover was increased (over 10-fold) by half-cutting on our study area. Guthery (1997) argued that usable space was limiting for northern bobwhites. Although distance to woody mottes limited habitat usability in his northern bobwhite model, Guthery (1999) suggested that availability of herbaceous land-cover also might be limiting. The height of the understory vegetation in our study was increased by 1.4 cm primarily due to the mechanical protection afforded by the half-cut branches from large herbivores. By cutting smooth-barked, multi-stemmed mesquite, the area of protection was increased >10 times. Because smooth-barked limbs are less brittle than rough-barked limbs, they survived better when cut and forced to the ground. Although we cut limbs that met all of the criteria suggested in Jackson (1969), our study suggested that one should concentrate on cutting the more tolerant smooth-barked trees (Jackson 1969, Webb and Guthery 1982). In cases where rough-barked trees are all that are available, cutting can still provide some protection although survivability of cut limbs is lower.

There was a positive response of quail food plants under the half-cut trees which was related to an increase in height and density of vegetative understory. Eight food plants responded positively to half-cutting, while 3 were negatively affected.

Management Implications

Our study suggests that half-cutting can be a valuable tool for increasing bobwhite use of areas that are lacking in cover. Half-cutting protects herbaceous vegetation from large herbivores, thereby creating islands of cover and additional quail food in grazed pastures. Management efforts should focus on maximizing habitat usability both spatially and temporally. We believe that half-cutting can add useable space on many heavily grazed rangelands.

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Testing the Value of Prickly Pear Cactus as a Nest-Predator Deterrent for Northern Bobwhite

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Recent research indicates that northern bobwhites (*Colinus virginianus*) in Texas commonly nest in prickly pear cactus (*Opuntia* sp.) instead of conventional bunchgrass habitat. We hypothesized that bobwhites nested in prickly pear because it served as a deterrent to nest predators thereby increasing probability of nest success (nest-protection hypothesis; Slater et al. 2001). We experimentally tested the nest-protection hypothesis by providing 50 wild-caught, captive raccoons (*Procyon lotor*) with combinations of simulated, bobwhite nests. Nest combinations included either 1 nest in bunchgrass (e.g., little bluestem [*Schizachryium scoparium*]) and 1 nest in prickly pear cactus (partial [75%] or full [100%] protection), or 2 separate nests in prickly pear (partial and full protection). Raccoons depredated 97%, 33%, and 14% of simulated nests constructed of bunchgrass (n = 35 nests), partial protection prickly pear (n = 30 nests), and full protection prickly pear (n = 35 nests), respectively. Prickly pear nests that provided full protection exhibited better survival against raccoon depredation than other nest types. Our study provides support for the nest-protection hypothesis regarding why northern bobwhite possibly nest in prickly pear cactus.

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Key words: cactus, *Colinus virginianus*, nest, northern bobwhite, predation, *Procyon lotor*, raccoon

Introduction

Northern bobwhites (*Colinus virginianus*) sustain a high incidence of nest failure, and depredation often is cited as the primary cause. To illustrate: Klimstra and Roseberry (1975) reported that, in Illinois, only 34% (n = 863 nests) were successful with predators accounting for 55% of nest failures; and in southern Texas, Lehmann (1984, p. 91) documented that 45% of 532 nests were successful, and depredation accounted for 84% of nest failures. In light of these high nest-depredation rates, researchers have speculated that nest depredation may be a limiting factor of bobwhite recruitment (Hurst et al. 1996, Rollins and Carroll 2001).

Probability of nest success can be influenced by various factors including nest location and nesting substrate. Research indicates that dense, residual cover can reduce nest depredation for various ground-nesting gamebirds (Schrank 1972, Kirsch 1974, Duebbert and Lokemoen 1976). Slater et al. (2001) documented that egg survival of simulated, bobwhite nests was proportionally related to density of potential bunchgrass nest sites. Martin and Roper (1988) hypothesized that predator efficiency decreased as the density of foliage surrounding the nest increased.

Bobwhites generally nest in bunchgrasses such as little bluestem (*Schizachyrium scoparium*) (Stoddard 1931, Klimstra and Roseberry 1975, Lehmann 1984, Peoples et al. 1996). However, in the southern Rolling Plains of Texas (Gould 1975), Carter et al. (2002) reported that 57% of bobwhite nests (n = 21) were located in prickly pear cactus (*Opuntia* spp.; hereafter, prickly pear) instead of traditional bunchgrass habitat. Hernandez et al. (2003) also reported that about 30% of bobwhites nests (n = 83) in this region were located in prickly pear despite adequate amounts of bunchgrass cover (>600 nest-

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ing sites/ha). More recently, Brooks (2005) provided further evidence of bobwhite nesting in prickly pear cactus in the Rolling Plains. Other instances of bobwhites exhibiting such a high use of prickly pear as nesting cover have not been reported in the literature (Lehmann 1984, p. 81). Slater et al. (2001) hypothesized that bobwhites were nesting in prickly pear because it provided mechanical protection against nest predators.

Given the recent documented use of prickly pear as nesting cover by bobwhites, we questioned why bobwhites used prickly pear as nesting cover. We developed a nest-protection hypothesis after Slater et al. (2001) and speculated that bobwhites nested in prickly pear because it provided nests structural protection against predators. The premise of the hypothesis is founded on the theory of natural selection, which implies that bobwhites will nest in areas that offer the greatest probability for success. Based on the nest-protection hypothesis, we predicted that survival of simulated, bobwhite nests would be higher for nests located in prickly pear than for nests located in bunchgrass.

Methods

We conducted an experiment to test the nest-protection hypothesis using 50 wild-captured, adult raccoons (Institutional Animal Care and Use Committee, Texas A&M University-Kingsville, No. 1-97-38). We captured adult raccoons in Kleberg County, Texas during October-December 1998. We selected raccoons as the nest predator because raccoons are considered to be the main predator of bobwhite nests in the Rolling Plains of Texas (Hernandez et al. 1997) where bobwhites have been documented to commonly use prickly pear as a nesting substrate.

We individually housed raccoons in kennels (1.2 m x 2.4 m x 2.2 m) and provided water and canned dog food *ad libitum* during a 2-3 day acclimation period. We then subjected each raccoon to 1 nest trial within their respective kennel. A nest trial presented raccoons with 2 simulated nests: either 1 nest in bunchgrass and 1 nest in prickly pear (partial or full protection), or 2 separate nests in prickly pear (partial and full protection). We used live prickly pear pads to construct a nest with either full protection or partial protection. Full protection was represented by 5 prickly pear pads that formed a cube around the nest with the ground representing the sixth side (Hernandez 1999). Thus, access to nests with full protection was obstructed by prickly pear pads from all angles. For partial protection, prickly pear pads formed 4 of the 5 sides of a cube, with the ground completing the cube (Hernandez 1999). Partial protection allowed access to nests from 1 side. To simulate a bobwhite nest in bunchgrass habitat, we cut and used bunchgrasses (e.g., little bluestem) to construct a nest bowl. All nests contained 1 chicken egg that was connected to a timer, which recorded the exact time of depredation (Hernandez 1999). We used a chicken egg instead of a quail egg because chicken eggs were readily available. We contend that using chicken eggs did not invalidate our study because once the protection afforded by the nesting substrate was breached, it is unlikely that egg size (chicken vs quail) would influence raccoon consumption of an egg in a captive setting.

We began nest trials at 1200 hrs and lasted for 24 hours. We withheld food, but not water, from the raccoons during the 24-hr period of the nest trial. We randomly assigned raccoons to nest trials. Twenty raccoons were used in nest trials consisting of simulated nests with full prickly pear protection and nests in bunchgrass. Fifteen raccoons were subjected to nest trials consisting of partial prickly pear protection and nests in bunchgrass. Lastly, 15 raccoons were used in nest trials consisting of 2 separate nests with full and partial prickly pear protection. We did not use in our nest trials raccoons that did not acclimate to the kennels (i.e., did not consume food or water during the acclimatization period). Our goal was to have at least 20 raccoons per nest trial, a logistically practical sample size given our captive facilities and the use of individual, wild-trapped raccoons for each nest trial. Unequal sample sizes arose because not all trapped raccoons acclimated to the captive facilities (e.g., too aggressive, never ate, etc.) and thus were not used in the trials.

We determined the depredation rank for each nest type by the order of depredation times. A nest type that was depredated first received a rank of 1; a nest type that was depredated second received a rank of 2; and a nest type that was not depredated within the 24-hr trial received a rank of 3. We used Kruskal-Wallis one-way analysis of variance by ranks test (Daniel 1987) to compare depredation ranks between prickly pear protection (full or partial) and bunchgrass nests, as well as between full protection and partial protection of prickly pear nests. We report all results as 0 ± SE and consider results significant at $\alpha = 0.05$.

Results

Nests in prickly pear with full protection survived for a longer period of time (0 survival category = 2.55 α 0.15) than nests in bunchgrass (0 survival category = 1.35 α 0.17; $P < 0.001$). Nests in prickly pear with full protection also survived longer (0 survival category = 2.53 α 0.16) than nests with partial protection (0 survival category = 1.45 α 0.21; $P = 0.0005$). However, there was no difference in survival time between nests in prickly pear with partial protection (0 survival category = 1.67 α 0.19) and nests in bunchgrass (0 survival category = 1.60 α 0.16; $P = 0.79$).

Considering nest survival for the entire study, 5 of 35 (14%) prickly pear nests with full protection were depredated during the trials compared to 10 of 30 (33%) prickly pear nests with partial protection. Thirty-four of 35 (97%) bunchgrass nests were depredated.

Discussion

The nest-protection hypothesis stated that nest success would be higher for nests located in prickly pear than for nests located in bunchgrass. Our data supported this prediction; nests with full protection were less vulnerable to raccoons than nests with partial or minimal protection. Our results were similar to Slater et al. (2001) who reported that simulated nests placed in prickly pear had a greater mean survival time than bunchgrass nests at sites with marginal nesting cover.

Our data suggest that prickly pear does not provide protection to nests in a binary manner, but rather the degree of protection appears to occur along a continuum. That is, the mere placement of nests in prickly pear does not guarantee protection against raccoons in an all or none manner, but rather protection spans between these 2 extremes depending on the degree of concealment. We documented that as degree of nest protection increased (from bunchgrass to partial to full protection), survival time of simulated nests also progressively increased. Hernandez et al. (2003) provided further evidence for such a trend under field conditions. They reported that 87% of bobwhite nests with full protection ($n = 15$) successfully hatched compared to 32% with partial protection ($n = 47$).

Although our data support the nest-protection hypothesis, we acknowledge that nest protection alone cannot explain why bobwhites are nesting in prickly pear. If nest fate depended solely on nest protection, then all bobwhites would be nesting in prickly pear because it represented the greatest probability for nest success during this simulated study. Naturally, this is not the case. Both Slater et al. (2001) and Hernandez et al. (2003) documented that nest fate was confounded with type of nesting substrate and surrounding bunchgrass density (i.e. overall nest concealment).

Hernandez et al. (2003) proposed that a new hypothesis that integrated the nest-protection hypothesis, and to a lesser extent a limited-bunchgrass hypothesis, could best explain why bobwhite nest in prickly pear. They reasoned that when bunchgrass cover became limited (e.g., following prescribed fire, overgrazing, or drought), prickly pear represented the most suitable nesting habitat (Soutiere and Bolen 1976, Carter et al. 2002). However, as bunchgrass cover increased, the probability of nest success equalized between prickly pear and bunchgrass habitat. Under these conditions, probability of nest success was similar between prickly pear and bunchgrass resulting in low or moderate use of prickly pear. However, this new hypothesis proposed by Hernandez et al. (2003) remains to be tested. A test of this new hypothesis could involve monitoring incidence of prickly pear use as a nesting substrate across along a continuum of study sites with increasing bunchgrass densities. Support for the hypothesis would detect an inverse relationship between prickly pear use and bunchgrass density as well as lower nest success for bunchgrass nests at lower bunchgrass densities but equal nest success between nest types (bunchgrass vs prickly pear) at higher bunchgrass densities.

We conclude that our study adequately supported the nest-protection hypothesis; however, it does not solely explain why bobwhites are nesting in prickly pear cactus.

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The Shepherd Project: a Case Study of Private Management for Ring-necked Pheasants (*Phasianus colchicus*) in Montana

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The Shepherd research farm is headquarters for Floating Island International, a company that produces floating wetlands for water quality and wildlife enhancement. It is also a demonstration site for managing ring-necked pheasants (*Phasianus colchicus*) on Montana landscapes. Primary goals for the 100-ha farm include an exploration of how agriculture can better integrate with wildlife to achieve a more sustainable landscape. One of the measures of this transition is ring-necked pheasant abundance. During 1998 to 2006, 3 management programs have been implemented: 1) changes in farming methods, 2) targeted habitat development, and 3) predation management. The main change in farming has been a reduction of irrigated annual cropland and pasture from 73% of the land area to 13%. The major crops planted on the site are more diversified and include corn, sorghum-sudan grass, barley, Maximillian sunflower, asparagus, and alfalfa. Other habitat management changes include delayed mowing and incomplete harvest of planted crops. During 7 years, the main predators removed by trapping were raccoon (*Procyon lotor*), striped skunk (*Mephitis mephitis*), feral house cat (*Felis domesticus*), red fox (*Vulpes vulpes*), and coyote (*Canis latrans*). Estimates of pheasant abundance were made during the hunting season using change-in-ratio of observed pheasants during male-only harvest. Each year harvest was halted when >75% birds observed were hens. Harvest using this method increased from 14 in 1998 to 207 in 2005. We believe this integrated approach, with its use of less intense agricultural methods, creates a diversified landscape that is beneficial to pheasants and enhances wildlife habitat for a variety of other species.

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Key words: habitat management, harvest, Montana, pheasant

Introduction

As managers of property which traditionally held populations of the ring-necked pheasants, but where harvest was generally much lower than we believe possible, we were interested in understanding what the level of sustainable wild pheasant harvest is on a farm managed intensively for wild pheasant. We hoped to gain an understanding of which variables were most important, relative to increasing levels of legal pheasant harvest on the study area. Similar measurement has taken place on larger study areas in Europe. For example, annual wild hen and rooster harvest at Seefeld Estate in Lower Austria demonstrated value of habitat management (Anderson 2002, Draycott et al. 2002, Bliss 2004). However, there are a number of key differences between these settings. Our goal was to track levels of sustainable wild bird harvest in a conventional farm ground setting in which certain potential variables could only be influenced within the relatively small study area.

Study Area

Our study area is located in Yellowstone County, about 30 km northeast of Billings, Montana. It is situated between 900 and 960 msl. The study area consisted of about 97 ha (240 acres) of farmland and associated habitats. During the 7 year study period, land use was transformed from mainly flood

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Table 1: Land Use at Shepherd Farm, Yellowstone County, Montana, before and after initiation of pheasant management program.

Description of Land Use	Land Use Prior to Study ha (acres)	Land Use Changes During Study ha (acres)
Flood irrigated annual crops	40 (100)	10 (25)
Flood irrigated pasture	30 (75)	12 (30)
Brushy draws and river bottom	25 (63)	26 (65)
Perennial crops (asparagus, Maximillian sunflower)	0 (0)	7 (18)
Ponds and wetlands	0 (0)	3 (8)
Fallow ground	0.4 (1)	34 (84)
Hedgerows	0.4 (1)	3 (8)
Orchard	0 (0)	1 (2)
Total	97 (240)	97 (240)

irrigated farm and pasture ground, to some native habitat, perennial crops and a mix of other habitats beneficial to pheasants (Table 1). Main annual crops are maize, sorghum-sudan grass and barley. Perennials include Maximillian sunflower, asparagus and alfalfa. Orchards are made up of cherry, plum and apple trees. An additional 7 acres of irrigated ground has been developed into weakly defined wind breaks and hedgerows during the study period. About 2.5 acres of the farm has also been transitioned into additional waterways that include ponds and ditches that flow consistently through spring and summer months.

Methods

Pheasant Harvest

During all 7 years of the project, pheasant hunting methods conventional to North America were utilized. Most hunts involved groups of hunters, typically 2 to 6, walking through cover with dogs, with some of the hunters strategically positioned in blocking positions that anticipated pheasant escape routes. Particular focus was placed on identification and recovery of any bird that showed signs of being hit by shot. This included loss of feathers, apparent flinching, or collapse of the bird to the ground.

Montana's hunting season ran for 10 weeks during the first 5 years of the study, then was lengthened to 12 weeks during the last 2 years. After the first month of hunting, hunters were consistently asked to report on the number of pheasants seen, and the ratio of hens to roosters. To ensure sufficient numbers of males relative to available spring breeding territories (Hill and Robertson 1988), hunting pressure was reduced once female to male ratio reached or exceeded 3 to 1 during the first 4 years of the study. After the first 4 years, pheasant numbers were estimated to be sufficiently high to ensure adequate roosters for available territories; therefore, hunting pressure was not reduced for the final 3 years of the study.

A pheasant survey was taken in late winter/early spring of each year of the study. A specific acreage of the study area would be pushed. Counters would tally roosters, hens and total pheasants seen. These results would be extrapolated over the rest of the study area to provide a generalised population count as well as hen to rooster ratios. We obtained harvest data for the rest of Yellowstone County, Montana for the first five years of the study. The percentage change in harvest was compared to the Shepherd project study area for all years data was available We then identified presence or absence of general foods found in each crop.

Habitat Management

In each study year but the first, a portion of annual farm crop was left standing, in some cases for the balance of the study. This took place with corn (maize), sorghum-sudan, annual and perennial sunflower, millet and barley. Up to 10 ha (25 acres) in a year were left unharvested and therefore available to pheasants and other wildlife on the property. At 2 times during the study 200-pound increments of medicated poultry grit were spread within high pheasant usage zones. Other than this, no other supplementary feeding was carried out in the study area.

The owner integrated a blended strategy to enhance for nest security. This involved coordinating land and water features so that security cover, food, (especially insect habitat associated with broadleaf plants) water and grit were present across the study area. "Edge" habitat was expanded by reduction in field size. Land disturbance was minimized during nesting season.

Except in one instance involving an aphid outbreak within a 0.6 ha (1 acre) orchard, pesticides or herbicides were not used within the study area. Fertilizer was applied sparingly, and at a rate approximately 2/3 lower than levels recommended for optimal agricultural production. Vinegar concentrate was utilized as an organic herbicide during the last 2 years of the study. Manure from a local feedlot was also spread through approximately 8 ha (20 acres) of the study area twice during the study period.

Alfalfa and grass hay were harvested from the study area intermittently through the study period, but never before July 15 in any year. Correspondingly, flood irrigation was also phased back. Currently, flood irrigation is initiated no earlier than the beginning of July. During nesting season and throughout summer months irrigation ditches provided a low volume source of water that presumably pheasant chicks could access and cross safely. Low water flow was maintained through these ditches that spread across the farm, providing a widely dispersed source of drinking water.

Predation Management

Several strategies were employed to manage small mammalian predators. Box traps armed with connibears were broadcast around the property and pre-baited, typically starting in November - December each year. At first, bait would be positioned in the box without the trap being set to condition predators to focus on the traps as food sources, then at the conclusion of the pheasant hunting season, the traps would be set. This was the primary trapping and predator control method utilized during the first three years of the study, and was effective at harvesting racoon, skunk and feral housecats.

As of year four a snare system was integrated into the predator management protocol, which was significantly effective in the harvesting of red fox, coyote and racoon.

As of year two, bait stations were also introduced into the predator control strategy. Carcasses and other odiferous materials were deployed in a brushy area, to attract predators. These locations, typically 2 on opposing sides of the research area, were then densely set with snares and box traps. Snares and box traps would otherwise be strategically positioned in funnel areas, and on or adjacent to deer trails within the property.

Hunting Method

During the first four years of the study hunters used shotguns with loads of their choice. During the 5th and 6th years, in most instances hunters were provided with shotgun loads of #4 or 6 Hevi-Shot. In year seven hunters were allowed again to shoot the load of their choice. In all instances, the study investigator accompanied hunters, unless it was determined that hunters were appropriately familiar with study protocols.

Crippled birds were verified on the following basis: as soon as possible after a shot sequence, hunters were queried about the disposition of the bird. In cases where the hunters indicated they hit a bird, but it was not recovered, hunters near the scene were

Table 2: Pheasant harvest at the Shepherd Project, Yellowstone County, Montana, during 1999-2005. This includes all legally harvested or wounded and lost males.

Year	Number Shot	Wounded and Lost	Recovered (%)
1999	14	3	11 (79)
2000	24	6	18 (75)
2001	41	11	30 (73)
2002	57	14	43 (75)
2003	92	21	71 (77)
2004	121	23	98 (81)
2005	207	21	186 (90)

asked for verification. If every hunter witnessing the episode confirmed that the bird had been significantly hit and was either going down or had gone down but had not been recovered, it was classified as a downed bird, in the "not recovered" category. A single hunter disputing this status would negate the entry. In cases without corroborating witnesses, hit birds that were not recovered were not tallied.

Results

Pheasant Harvest

Pheasant harvest has steadily increased from a low of 14 shot in 1999 to 207 shot in 2005 (Table 2). The percentage of shot birds that were recovered ranged from 73% to 90% with the highest recovery during the last 2 years (Table 2). Compared to slightly declining trend in harvest for Yellowstone County, there was a 1,400% increase in pheasant on the Shepherd study area (Figure 1).

Pheasant harvest increased 15-fold over the seven year study.

Pheasant Abundance

Post-hunting season pheasant population surveys were conducted for Year 2 and Year 7. The one-day surveys were conducted by flushing and counting pheasants from the areas of the property where most of the pheasant population was believed to reside. During February of the second year, 49 pheasants were counted, comprising 24 roosters and 25 hens. During the prior season 24 roosters had been shot. Despite this, hens to roosters were still 1:1, indicating that hens were experiencing similar mortality levels that year.

During March of the seventh year, 260 pheasants were counted, comprising 205 hens and 55 roosters. A territory count taken later that spring found 30 territories occurring in the study area, indicating 1.8 roosters available per territory, with 6.8 hens per territory, assuming no dispersal. Since dispersal is likely due to proximity of additional appropriate habitat adjacent to the study area, the hens-per-territory ratio is likely to lower somewhat. With a 205 hen count a further increase in next year's pheasant population seems probable.

We found a diversity of plant and animal materials in the crops of harvests birds (Table 3). The main animal food was grasshopper (Orthoptera). Most of the plant materials were seeds of corn, Russian olive, sorghum-sudan, and rose hips (Table 3).

Predation Management

We estimate that predator control activities consumed an average of 100 hours per year. Use of bait stations and funnel zones, quickset snares and pre-baited box traps on a small farm made the predator work significantly more effective, especially in relation to travel time between trap sets.

The most common predators trapped during predator removal were raccoons and striped skunks

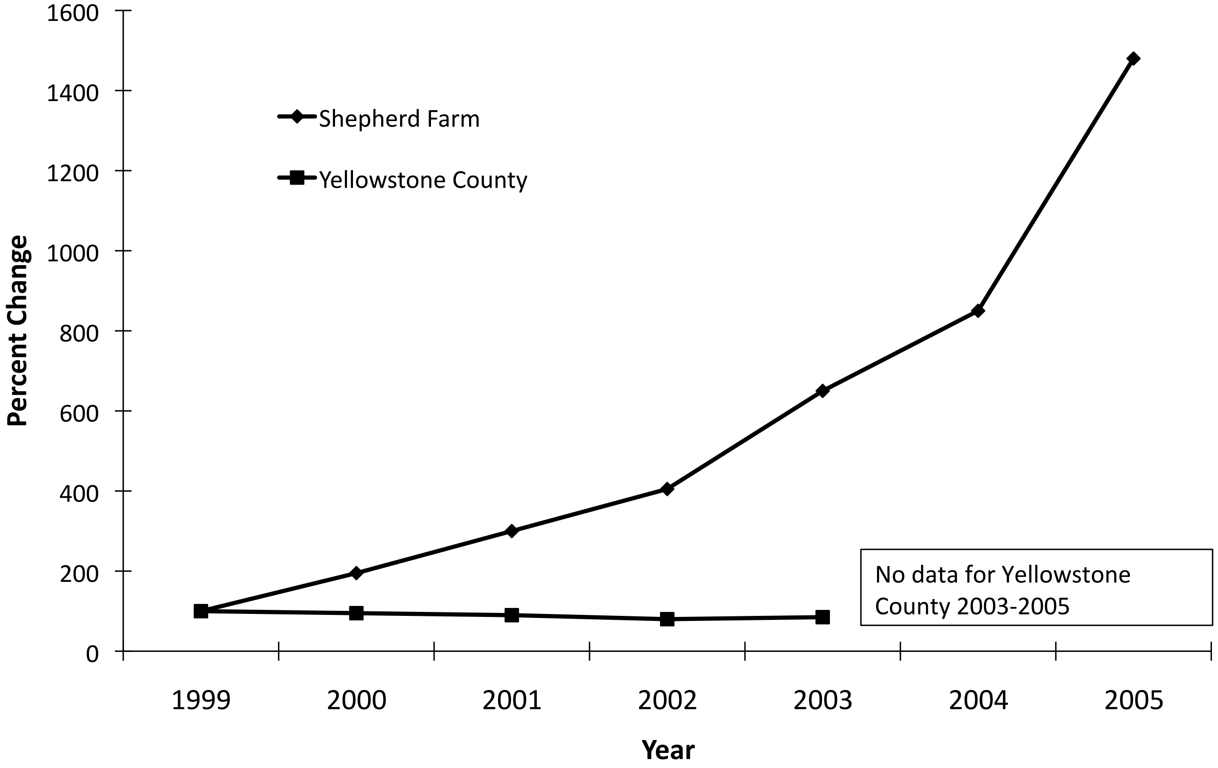

Figure 1: Change (%) in pheasant harvest on the Shepherd Project study area compared to the harvest from Yellowstone County, Montana during 1999 to 2005.

(Table 4). Large numbers of feral cats were also trapped on the study area. During the last year of trapping, mean weight of racoons harvested on the study area was 4.1 kg (9 lbs) compared to the 6.4 kg (14 lbs) average observed by a local fur buyer. For red fox we observed a mean weight of 4.3 kg (9.5 lbs) compared to the fur buyers reported 5.0 kg (11 lbs) average.

Discussion

The Shepherd research farm is headquarters for Floating Island International, a company that produces floating wetlands for water quality and wildlife enhancement. The company's production headquarters are located on a property adjacent to the research farm. Primary goals for the research farm include an exploration of how agriculture can be synchronized with wildlife to achieve an optimal and sustainable balance. The owner has determined that increasing organics in the soil and increasing the amount of land area planted in perennials compared to annual crops will contribute to the long-term goal. Because of the well-established research protocols and methods associated with pheasant, the owner chose to use them as one of the "flagships" tracking progress towards these goals.

The project modified 3 major variables that would positively impact pheasant abundance, farming methods, habitat, and predator demographics to achieve a 15-fold increase in pheasant harvest. Achieving this on a relatively small area was challenging. Pheasants could readily avoid the significant hunting pressure by moving to adjacent properties (the farm was hunted on average twice per week during the last two years of the study). On the other hand the 100 ha (234 acre) size presented a much more focussed opportunity to concentrate predators.

Increases in habitat dovetailed with the farm's

Table 3: Crop contents of harvested male pheasants on the Shepherd project study area, Yellowstone County, Montana during 1999-2005

Year	Contents of Crops
1	grasshoppers (Orthoptera), Russian olive (*Elaeagnus angustifolia*), Rose Hip (*Rosa* spp.)
2	corn (maize), Russian olive, sorghum-sudan
3	Russian olives, sorghum-sudan, corn (maize)
4-7	corn (maize), Russian olive, sorghum-sudan

goal of expanded perennial cover. The owner estimates that between 6,000-7,000 trees and bushes were planted during the study period. It is also noteworthy, however, that these plantings are not likely to have had much bearing on study results for 2 reasons: drought conditions, and unwillingness on the owner's part to utilize chemicals for weed control. As a result tree and bush survival and growth have been low to moderate. What has contributed to enhanced brood survival, the owner believes, is the strategy of not harvesting up to 10 ha (25 acres) of crop per year, and leaving these crops, primarily sorghum-sudan grass and corn, standing or lodged over for up to 5 years, with corresponding broadleafed weeds like koshia (*Koshia scoparia*), Russian pigweed (*Axyris amaranthoides*), white or yellow-blossom sweet clover (*Melilotus officinalis*) and mustard (*Sinapis arvensis*) filling in and providing dense security cover and thermal mass. Integrating low water volume ditching with these weedy patches, as well as some manure strips, akin to the European beetle bank strategy, provided for added pheasant chick survival enhancement (MacLeod et al. 2004).

We have refrained from burning on the property, but we have used cattle to graze off major areas of the farm twice during the study period, both times during winter months. Some fields have also been mowed in late winter/early spring, then tilled into the ground. After testing for mycorrhizial presence, selected sites around the farm have been inoculated with a commercial blend of micorrhizia. The farm has also incorporated approved insect vectors into its weed control strategy in order to meet mandated weed control guidelines.

As explanation for the significant improvement in the recovered birds ratio in the last year of the study, the owner proposes the following theory. Montana had been in a long-term drought throughout the study period. Higher than normal precipitation occurred during the last year of the study: fall precipitation was double the previous year. Added moisture seemed to enhance the dogs' ability to find downed birds. In addition, the ample moisture seemed to result in many high quality shot opportunities as birds would hold tighter in the more dense cover. We propose that these 2 considerations combined to provide a setting that compared to more typical Midwestern cover and moisture conditions, and to recovery ratios that have been reported from that region. It is also noteworthy that Hevi-shot, while not mandated for the last year of the study, was the predominant load selected by hunters and is also the year with the highest recovery rate of shot birds.

Over the next several years we intend to expand wetland habitat on the property. Expansion of optimal habitat in concert with conscientious farming methods and predator control could result in further expansion of the pheasant harvest.

Of the 3 variables - habitat improvement, predator management or adjusted farm practices we believe that all 3 are important and actually become syngernistic in their value. Additional research into various factors would certainly be of value, for

Table 4: Predator removal from the Shepherd project study area, Yellowstone County, Montana, during 1999-2006.

Species	Number Harvested
Raccoon	279
Red fox	78
Coyote	40
Striped skunk	117
Feral cat	111
American mink	4
Long-tailed weasel	2

example, into the impact on pheasants and other ground nesting birds of pesticide and herbicide use; into variations in predator effectiveness relative to predator age and experience and into tracking effectiveness of weed-infested standing crops as a pheasant enhancement strategy.

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Impact of Game Crops Intensification and Hunting Management on Red-Legged Partridge

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In the French Mediterranean region, large cultivated areas have been abandoned. Matorrals (shrublands) then woodland successional stages replace the mosaic of habitats, favourable to the red-legged Partridge *Alectoris rufa*. An agro-environmental operation, consisting of development of areas devoted to game crops, allowed the reopening of these landscapes on the studied site. This we followed by the development of a hunting plan and operation. We tested the impact of these managements on the dynamics of the red-legged partridge abundance. Our research showed that the limitation of hunting bag has a strong impact on the densities of partridge in spring, whereas habitat managements act favourably on the success of the reproduction and recruitment.

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Key words: *Alectoris rufa*, breeding success, brood size, game crop, habitat management, hunting management, French Mediterranean region, red-legged partridge, population dynamics, rural abandonment

Introduction

Agricultural declines have had a deep impact on the Mediterranean region of France: the progressive abandonment of vineyards and olives, pastures or gathering in scrubland, combined with rapid urbanization, has gradually transformed the Mediterranean landscape. This modern landscape, primarily comprised of mosaics - hills or plains, has resulted in matorrals, forests and fallow lands has extended, reducing habitats favourable to the species related to the open landscapes or edges for typically Mediterranean species, such as red-legged partridge. This important game species is representative of the Mediterranean landscape and is an important economic and conservation flagship of the region. These extensive modifications of its habitat constitute certainly one of the causes of its decline (Aebischer and Potts 1994, Ponce-Boutin et al. 2003*b*).

For the maintenance of partridge populations, various techniques of management have been developed, but more hill (Ponce-Boutin et al. 2003*a*) rather than in plain landscaped (present study). The objective of this project was to test if management of abandoned vineyards could influence the population dynamics of red-legged partridge. In addition we tested the effects of a hunting management program.

Study Area

Located in the Mediterranean area of southern France, the commune of Pailhès (600 hectares) was almost exclusively a wine making area. From 1975, subsidies were distributed in order to decrease production of lower quality wines. In 1994, at the beginning of this study, the vineyard accounted only for 71% of the agricultural area. Much of the abandoned vineyard was in various stages of plant succession, including waste lands, matorral (15%) or wood (2%). Finally, 6% of the abandoned grounds remained cultivated, primarily with cereals.

Methods

The farmer's association of Herault, the Hunters Departmental Federation of Herault and the Na-

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tional Hunting and Wildlife Agency joined in 1994 to work on a project of agro-environmental measurements. The project was divided into contracts, helped over 5 years and suggested to farmers the creation of game crops (with sowing every 2 or 3 years), or the maintenance by crushing in some areas of 13 communes. For the commune used in this study, Pailhès, management was only applied to approximately 200 ha, the other part remained as test-control (363 hectares). The remaining area was occupied by the village and some private properties. Land use was controlled on its whole territory. 48 parcels thus were the subject of a contract, a total of 15.1 hectares, distributed on the managed portion. Four time periods created based on habitat managements in place: a first phase prior to the implementation of the measurement prior to 1994, a second phase of management implementation (1994 to 1998), a third phase of transition (1999-2000), where the majority of the game crops were converted into corn (Figure 1), and finally a fourth phase, since 2001, when no management was taking place.

In addition, as of the hunting season 1994-95, a hunting plan (limitation of the number of partridges to be hunted) was developed and where reductions of harvest were reduced to a take of 4 partridges per day. It was established in its final version from the autumn 1996. We distinguished 3 phases: no bag limit (hunting seasons 1992-93 and 1993-94), followed by the phase of the hunting plan set up (seasons 1994-95 and 1995-96), and finally implementation phase of hunting management starting in autumn 1996. A sector of 55 ha within the control area remained a hunting preserve throughout the period.

The number of partridges present in spring on each of the three zones was estimated annually during 1993-2006 (except 2000 and 2005) by using 9 drive counts. Reproductive success was estimated each year between 1994 and 2005 by the number of broods per adult and the average size of the broods on the managed and control area (except reserve) by means of the method of sampling of coveys (Office National de la Chasse et de la Faune Sauvage 2004).

We tested the impact of the various management techniques, applied year N, on the density and the success of the partridge reproduction occurring the year N+1. The analyses consisted in successively testing the effects "hunting management" and "habitat management" by a univariate general linear model (two-way ANOVA) for each variable. In order to obtain the homoscedasticity, the variable density adjusted using a square root transformation.

In the experimental plan (figure 2), an observation represents a sector (managed area, control area out of reserve, reserve) x year. We used the measure of balance for one-way design $y = \tilde{n}/\bar{n} \leq 1$ where $\tilde{n}$ and $\bar{n}$ denote the arithmetic and harmonic means of the group sizes, respectively (Ahrens and Pincus 1981, Ahrens and Sanchez 2006). $y = 1$ if the design is balanced; the closer y is to 0, the more unbalanced is the design. We found that all the one-way designs are only slightly unbalanced ($y < 0.83$). Nevertheless, following Shaw and Mitchell-Olds (1993), when computing the two-way ANOVA with habitat management and hunting management as factors we used the so-called Type III sum of squares for performing the tests. ANOVA computations were done using SPSS 14.0 (SPSS Inc., Chicago IL).

Results

Spring density of red-legged partridges

The abundance estimates in spring of red-legged partridge populations revealed a total of 1245 birds during 1993 and 2006.

Intensification of game crops surfaces did not have a significant effect on the mean density in spring (F = 0.526, 2 df, P = 0.60), contrary to hunting management (F = 5.968, 3 df, P = 0.003). Indeed, following the installation of a bag limit, the partridge density doubled (increasing from 6.7 pairs/100 ha to 13.6 pairs/100 ha (Table 1). On the reserve, the average density appeared intermediate.

Number of broods per adult in summer

Surveys carried out during 1994-2005 outside of the hunting preserve revealed a total of 395 broods for 1198 adults.

It seems that neither the mode of hunting management (F = 1.827, 2 df, P = 0.19), nor the habitat managements carried out on the landscape (F = 1.768, 2 df, P = 0.20) influenced the average number of broods observed per adult (0.35 ± 0.13 SE).

Average brood size in summer

We recorded 412 broods throughout the study, representing a total of 2,499 chicks observed.

Bag limit did not seem to have a significant effect on the average size of the broods (F = 1.345, 2 df, P = 0.28). Conversely, a habitat management did have an impact on brood size (F = 4.133, 2 df, P = 0,032). Indeed, following the intensification of the game crops surfaces, average number of young partridges per brood increased by 30% (Table 2).

Discussion

The sowing of new game crops done within the framework of agro-environmental measurement does not appear to influence spring density of red-legged partridge. Conversely, as was shown by Ponce-Boutin et al. (2006), the development of a hunting plan on the commune, using bag limits, had a clearly positive impact on the abundance of the species.

It is possible that the influence of the cultures on this factor was hidden by the shooting carried out on partridges. Indeed, this one follows from the proposals calculated from the densities recorded in spring and from the success of the reproduction in the summer preceding the hunting season. The number of produced young increases all the more the bag suggested, what tends to level the number of birds next spring.

However, it appeared that the installation of cultures instead of closed landscape such as matorral, waste lands and woods, seemed to have a positive impact on the size of the broods. In order to confirm this study, the experiment was maintained on the same commune by reversing the role of each sector, i.e. by establishing game crops on the sector initially in control. Jointly, the hunting plan continues.

Additional studies undertaken on these populations suggest that the fecundity of hens is not in question, but rather the survival of the young partridges in their first weeks of life (F. Ponce-Boutin, unpublished data). It should indeed be considered the fact that surveys done during summer to estimate the success of the reproduction are carried out 6 weeks after peak hatch, i.e. once the young birds underwent the most extremely of mortality rate. These game crops seem essential for chicks because they would offer to partridges, sufficient food as well as cover protecting them from predators, allowing a better survival.

Management Implications

Within sight of the results of the diet studies on red-legged partridges, the optimum seems to be a mixture including at least a cereal for the seed contribution in summer and of leguminous plants for the insects, nitrogen which they fix and the cover they offer in addition of food (Bro and Ponce-Boutin 2004, Green 1984, Ponce 1989). These sowing could also be laid out in inter-rows in orchards or vineyards (see Borralho et al. 1999) or being established on the set-aside lands, of which impact on the more effective in the countries with less-intensive agricultural practices and higher fractions of land removed from production (Buskirk and Willi 2004). It would be necessary to seek a uniformed distribution on the territory in order to make them available to a maximum of broods. It is not recommended crushing them or making them grazed in spring or in summer; on the other hand, maintaining a pasture of maintenance apart from these periods can only be favourable (nitrogen contribution by dejections, limitation of the progression towards waste lands, matorrals or woods). Finally, they will not make objects of any weed-killer treatment or pesticide.

Habitat management must be implemented jointly with limitations on the hunting bag so as to make it possible the partridge populations to reach their optimum level of density. The higher those will be, the more the effect of the increase of success of the reproduction will be noticeable.

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Effects of Two Burn Scales on Northern Bobwhite Demographic Parameters and Home Range Size

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Properties managed for bobwhite hunting in the Red Hills region use prescribed fire to burn 40-60% of their upland areas annually. Burning large areas could negatively affect bobwhite survival by increasing predation risk. Burning small areas could influence daily habitat use patterns and may influence nesting or brooding processes. Research on how the scale of management influences home range size and demographics of bobwhites is lacking. Therefore, on Tall Timbers Research Station (TTRS) we established 4-140 ha experimental treatment areas and randomly assigned 2 areas to be burned at a small scale (∼2.25 ha burn patch size) and 2 areas to be burned at a large scale (∼8 ha burn patch size). Upland habitat areas were divided into similar size patches, depending on treatment, with half of the units in the treatment areas burned annually in an alternating pattern. We annually radio-tagged bobwhites (2003-2005) and monitored their survival, productivity, and home range size during the breeding season. In 2003-2004, the risk of mortality was higher on large scale versus small scale treatments during March-May, but there were no differences in 2005. Per capita nest production was higher on small scale areas (0.68 nests/bird) than large scale areas (0.43 nests/bird). Autumn density was higher on small scale treatment areas (3.5 bobwhite/ha, SE = 0.62) compared to large scale treatment areas (2.7 bobwhite/ha SE=0.67; $F_{1,2}$ = 12.9, *P* = 0.07). On TTRS, there were marginal benefits of small scale management for early breeding season survival and productivity, especially during years with high late winter mortality or slow vegetative growth after burning due to drought, but the gains we observed may not be enough to supersede other management priorities.

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Introduction

Prescribed burning is an important management practice for southeastern hunting plantations managed for northern bobwhite (*Colinus virginianus*; hereafter: bobwhite; Stoddard 1931, Rosene 1969). Prescribed burning improves habitat for bobwhites by increasing bare ground, increasing availability of seeds and insects when foraging, and promoting herbaceous plants and shrubs important for cover and food (Brennan et al. 2000). To achieve these benefits, southeastern hunting plantation managers typically use prescribed fire on a regular 1-2 year interval on 40-70% of their uplands each year, a frequency and extent that is necessary to maintain upland habitat conditions suitable for bobwhites. However, the scale of burning differs; some managers burn upland areas in patch sizes of 8-20 ha (20-50 acres) or greater, while others protect small patches 0.5-2.5 ha (1-5 acres) scattered throughout the upland landscape to create a mosaic of burned and unburned habitats (Rosene 1969, Landers and Mueller 1986). Burning upland habitats at these different scales creates a matrix of different habitat conditions and varying amounts of edge between burned and unburned habitats. These varying patch configurations and the usability of these patches through time could influence important demographic parameters such as survival, productivity, and home range sizes that could influence autumn population size, the most important population variable for a hunting plantation.

Prescribed burning also can increase the short-term susceptibility of bobwhites to predation by removing protective cover and forcing movements to

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and use of unfamiliar areas (Stoddard 1931, Mueller and Atkinson 1985, Folk 2006). The predation risk for bobwhites could likely increase as the burn area increases in size. However, if adequate cover exists within a reasonable distance or within a home range of a bobwhite, survival impacts may be minimized. Additionally, the scale of burning or patch size along with the usefulness of those patches can influence susceptibility of adults, nests, and broods to predation. As patch sizes decrease, the ability of predators to effectively forage and find prey may increase (Jimenez and Conover 2001, Wiens 1989). The effects of burning at different scales may create competing costs and benefits when managing for bobwhite abundance.

There is little information on the direct effects of the scale of burning on bobwhite survival and productivity on southeastern hunting plantations. Therefore, our objective was to compare the effects of burning at a 2 ha scale and 8 ha scale on bobwhite survival, reproductive response, home range size, and autumn bobwhite density.

Study Area

We conducted this study on Tall Timbers Research Station (TTRS; 1,568 ha), Leon County, Florida. The landscape composition of TTRS was rolling hills consisting of primarily upland pine forests (66%) including longleaf pine (*Pinus palustris*), loblolly pine (*Pinus taeda*) and shortleaf pine (*Pinus echinata*) in the overstory, and equal proportions of grasses, forbs, and shrubs associated with "old-field" plant succession in the understory. Intermixed throughout the uplands were hardwood drains and hammocks (21%) and annually disked fallow fields (13%) 0.4 to 1.2 ha in size. Prescribed fire was used throughout TTRS on a 2-year interval to control encroachment of hardwood trees and to maintain the mix of herbaceous and woody ground cover. Other management practices used in the uplands were mowing and roller chopping of hardwood and pine saplings.

As part of a separate long-term research project, TTRS was divided into 2 sides with 1 side receiving supplemental feed. Approximately 5 liters/ha of grain sorghum were spread twice monthly along designated feed trails through the uplands. Additionally, in 2004-2005, we implemented mesomammal predator reduction on the entire property between March and October, which consisted of removal of raccoons (*Procyon lotor*), opossums (*Didelphis marsupialis*), armadillos (*Dasypus novemcinctus*), bobcats (*Lynx rufus*), coyotes (*Canis latrans*), and gray foxes (*Urocyon cinereoargenteus*; University of Georgia, IACUC #: A2001-100100-0).

Methods

Scale Treatments

We selected 4-140 ha areas on TTRS with similar landscape composition; 2 areas had supplemental feeding and 2 areas had no additional feeding. Areas were randomly assigned either a large scale treatment, for which upland habitats were divided into 8 ha blocks, or a small scale treatment, for which upland areas were divided into 2.25 ha blocks (Figure 1). In 2002, on all treatment areas, blocks were selected in an alternating pattern and were burned March-April, such that half of the uplands were burned. The following year the unburned upland blocks from the previous year were burned to create a 2 year burning rotation for all upland areas. This alternating burn pattern was continued during the subsequent years of the study (2003-2005).

Telemetry Data

Bobwhites were captured in January and March using standard walk-in funnel traps (Stoddard 1931). We determined sex, age class, and weight for each captured bobwhite and attached a uniquely numbered aluminum leg band (National Band and Tag Co., Newport, KY 41072). From each captured covey we selected 2-3 bobwhites to be fitted with a 6-g radio transmitter (American Wildlife Enterprises, Monticello, FL 32344). Radio transmitters were distributed at an approximate ratio of 1 radio-tagged male to 4 radio-tagged females. In January, trapping efforts and radio distributions were equally applied to all areas of TTRS, while in March trapping was

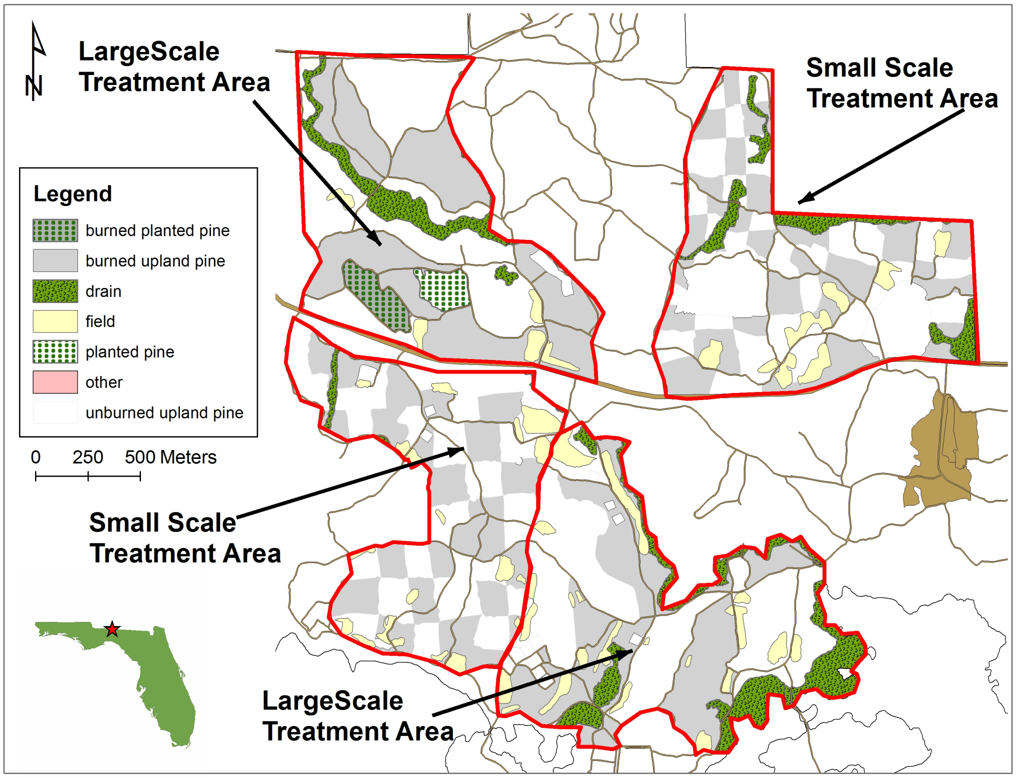

Figure 1: Small scale (2.25 ha) and Large scale (8 ha) treatment areas on Tall Timbers Research Station, Florida, USA, 2003-2005.

focused on areas with an inadequate radio tag sample. Trapping, handling, and marking procedures were consistent with Palmer and Wellendorf (2007) and followed the guidelines of the Tall Timbers Research, Inc. Institutional Animal Care and Use Committee Permit (#GB2001-01).

Radio-tagged bobwhites were located at least once weekly until 15 April and then 3-5 times a week thereafter until 1 October. We determined locations of radio-tagged individuals using homing procedures (White and Garrott 1990) and then marked locations on detailed landcover maps developed in ArcGIS (ESRI, Redlands, CA). While the precision of calculated locations to the actual locations of radio-tagged bobwhites has not been formally determined, we thoroughly trained technicians on the homing technique to ensure they determined locations to within at least a 30 m^2 area. Additionally, they verified that the correct macrohabitat landcover type (e.g., burned upland, unburned upland, field, hardwood drain) was assigned to the location.

Transmitters motionless for 12 hours changed pulse rate to notify observers of a potential mortality. Idle transmitters were located to determine if mortality had occurred or if the transmitter had slipped off of the bobwhite. All speculated radio slip observations and failed transmitters were censored from the survival analysis.

We determined that a bobwhite was nesting when its locations were unchanged for 2 consecutive days and the mortality sensor had not activated. Nesting sites were flagged and monitored daily until a depredation or hatch. Bobwhites with a hatch were treated as a brood until 14 days post hatch or until it was determined the bobwhite was no longer with a brood. We assumed that we detected all incubated nests of radio-tagged bobwhites for the productivity index, which is plausible since bobwhites were located frequently during the breeding season.

Autumn Covey Density

We conducted morning covey call surveys September-November. Three 25-ha square survey quadrants were randomly located on each treatment area. An observer was located on the mid-point of each quadrant side, totaling 4 observers per quadrant. From 45 minutes before sunrise until sunrise, observers recorded bearing, estimated distance, and plotted locations of all calling coveys on maps within the quadrant and surrounding areas. All observers had practiced collecting data at least twice before participating in the surveys, and most of them had extensive experience from surveys of previous years (Wellendorf and Palmer 2005). We determined calling covey locations via biangulation or triangulation by two or more observers or locations verified by a single observer within 100 m of the calling covey. This process gave us an accurate count of all calling coveys within the 25 ha survey area and we assumed all calling coveys were accurately detected.

We calculated bobwhite density within the 25 ha area by adjusting the calling covey count by a predicted estimate of the calling rate (Wellendorf et al. 2004). This adjustment produced an estimate of the number of coveys within a grid, which we then multiplied by the average covey size from flush data collected over the entire treatment area. We used an average covey size since it was not possible to consistently flush all coveys or all bobwhites within a covey out of a particular 25 ha survey area.

Data Analysis

For the survival, productivity, and home range analyses individual radio-tagged bobwhites were assigned a treatment classification (small or large scale) using telemetry locations. Within each year, bobwhites were assigned a particular treatment if more than 70% of all locations were within the boundaries of a treatment area. This arbitrary assignment yielded a dataset of radio-tagged bobwhites most affected by the treatment area where the majority of its time was spent, but did not restrict inclusion of radio-tagged bobwhites that occasionally went outside of the treatment area boundaries.

Survival - Given the limited sample sizes of radio-tagged bobwhite on some treatment areas for individual years and the potential biases associated with Kaplan-Meier product limit method (Kaplan and Meier 1958) when using small sample sizes (Pollock et al. 1989, Palmer and Wellendorf 2007) we selected Cox's proportional hazard modeling (Cox and Oakes 1984, SAS Institute, Inc. 2002, PROC PHREG) to investigate differences in the hazard rates for scale treatments (small, large), side (fed, not fed), and year (2003-2005). The hazard rate model relates failure times (death or censored), which is the equivalent to the instantaneous mortality rate, to the explanatory variables. We analyzed the early breeding season hazard rate, designated as 1 March-1 July, for each year. We hypothesized that burning at different scales would have the greatest effect on survival during this time period. We treated each year as independent due to the amount of time between breeding seasons and the environmental differences among years. Preliminary analyses tested for time dependence within the breeding season for each explanatory variable in the model; if we detected no time dependence for an explanatory variable, we used the main effects explanatory variable.

Productivity - We calculated 3 productivity parameters; a per capita female productivity index, female nest success, and a per capita female hatch index for each scale treatment each year. Observations were pooled over both sides due to limited sample sizes. The female productivity index was calculated dividing the total number of female incubated nests for a given breeding season by the number females entering the breeding season. Radio-tagged bobwhites were considered part of the breeding population if they were alive on 15 April of each year. Nest success was presented as a percentage of successful nests. Finally, we calculated the female hatch index, which the productivity index multiplied by nest success. Nests abandoned due to observer intrusion were censored from nest success calculations and the hatch index. Male nest incubation was not investigated due to limited sample sizes.

Autumn Density - Survey grids were sampled once and were considered sub-sample observations used to estimate a covey density treatment mean for each year. Data were analyzed using a 3 factor ANOVA (Stat Soft Inc. 2001) with 2-way interactions for the variables: scale, side, and year. Each year was treated as independent. Small sample sizes prevented us from conducting repeated measures analysis for each treatment area by year. Even though there was some unexplained variation in density associated with each specific treatment area we presume that it would have minimal effects on the treatment results.

Home Range Analysis - Early breeding season (1 March-1 July) home ranges were calculated for all radio-tagged bobwhites surviving until 1 July. We used surviving individuals because we wanted to investigate treatment differences in home range size over the entire period of interest, and we did not want the analysis biased by premature deaths or small sample sizes. We used all locations, including locations collected during brood-rearing and one nest site location, for home range analyses. We estimated 95% fixed-kernel home ranges using a fixed smoothing parameter (H_{ref}) estimate of 38, which was an average of the least squares cross-validation (LSCV) H_{ref} values for all radio-tagged bobwhites during the breeding season, 2003-2005 (Kenward 2001). Home range estimates were calculated for each scale treatment each year, but were pooled by side, due to sample size concerns. Additionally, each year was treated independently due to the environmental differences between years and the length of time between calculations of seasonal home ranges.

Results

For the hazard rate modeling analysis we monitored 33 radio-tagged bobwhite in 2003, 33 in 2004, and 46 in 2005 on the small scale treatment areas, and on large scale treatment areas we monitored 108 radio-tagged bobwhite in 2003, 40 in 2004, and 49 in 2005. We observed no statistical differences in the hazard function or instantaneous mortality rate by side for any year (Wald $\chi^2 > 1.91$, DF = 1, $P > 0.166$). The effect of scale on the hazard rate varied by years. We found no significant effects (Wald $\chi^2 > 1.53$, DF = 1, $P > 0.215$) of scale on hazard rate in 2003 or 2005. However, in 2004 bobwhites on small scale treatment areas were 2.3 times less likely to die than bobwhites on large scale treatment areas (Wald $\chi^2 = 7.91$, DF = 1, $P > 0.005$). While the scale treatment was not significant in 2003 there was a trend of a lower hazard rate in the first 60 days of the breeding season for small scale areas.

The number of radio-tagged female bobwhites alive on 15 April was 25 in 2003, 22 in 2004, and 28 in 2005 for small scale areas; on large scale areas there were 47 in 2003, 25 in 2004, and 33 in 2005. We documented 94 incubated nests for all years and treatment areas, of which 6 (6%) were abandoned due to observer intrusion (Table 1). In all years, we recorded higher nest productivity on small scale treatment areas (1.4-1.5 times larger; Table 1). However, in 2003 and 2005 we observed slightly higher nest success on large scale than on small scale treatment areas, 1.4 times larger in 2003 and 1.2 times larger in 2005. Results were varied for the female population hatch index. In 2003, hatch index was similar by scale treatments, but in 2004 and 2005 the hatch index was 1.5 and 1.3 times greater on small scale than on large scale treatment areas.

Autumn bobwhite density was 2.9 ± 0.6 (SE) bobwhite/ha in 2003, 2.5 ± 1.0 in 2004 and 3.9 ± 0.8 in 2005. We observed a significantly larger autumn population on the fed side (4.33 ± 0.48 bobwhite/ha) than on the unfed side (1.87 ± 0.47 bobwhite/ha; $F_{1,2} = 18.18$, $P = 0.008$). Additionally, there was a trend for higher autumn bobwhite density on small scale treatment areas (3.51 ± 0.77 bobwhite/ha) than on large scale treatment areas (2.7 ± 0.82 bobwhite/ha; $F_{1,2} = 12.89$, $P = 0.07$), but the degree of difference between treatments was variable among years (Figure 2).

We calculated March-June home ranges for 121 radio-tagged individuals, 2003-2005. The mean

Table 1: Sample sizes for radio-tagged female bobwhites alive on 15 April and their reproductive output on small and large scale treatment areas on Tall Timbers Research Station, Florida, USA, 2003-2005.

		Radio-tagged Females	Female Nests	Productivity Index[a]	Female nest success (%)	Hatches/female[b]
2003	LARGE	47	15	0.32	60	0.19
	SMALL	25	11	0.44	42.9	0.19
2004	LARGE	21	13	0.62	66.7	0.41
	SMALL	22	21	0.95	66.7	0.64
2005	LARGE	33	15	0.45	85.7	0.39
	SMALL	28	19	0.67	73.7	0.5

[a] Total number of incubated nests for a breeding season / female radio-tagged sample alive on 15 April

[b] Total number of female incubated nests * female nest success / female radio-tagged sample alive on 15 April / 100

number of radio locations used per individual was 49.1 (SE = 1.6; range: 18-92). We observed a significant interaction between side and year ($F_{2,111}$ = 180.21, P = 0.01), but observed no significant differences in home range size by scale ($F_{1,111}$ = 5.64, P = 0.58). However, we did observe a trend for larger home ranges on large scale treatment areas in 2004 (Table 2).

Discussion

We observed that the risk of mortality was marginally higher on large scale treatment areas during the early part of the breeding season for 2 of the 3 years. This result may have been influenced by avian predator migration patterns and abundance during the early breeding season (Mueller et al. 1988). Annual survival in 2003 and 2004 was lower than in 2001 and 2002 on TTRS (Palmer and Wellendorf 2007), which was attributed to greater proportions of avian kills of radio-tagged bobwhites during the early breeding season on TTRS (Faircloth et al. 2005, W. Palmer and S. Wellendorf, unpublished data). Radio-tagged bobwhites were observed using small patches of residual cover in burned areas soon after a fire, which is similar to findings of Mueller et al. (1988) and Carter et al. (2002), and therefore may have been more susceptible to avian predation on large scale areas due to limited proximity of thicker protective cover. An additional factor potentially increasing mortality risks to bobwhites was a decrease of cotton rat abundance, a common alternative prey (Schoch 2003), which was lower on large scale than small scale areas during the early breeding season for all years (Hannon 2006). In 2004, we observed the largest difference in mortality risk between treatments, which was associated with a below average rainfall in March and April of 2004 (Nation Weather Service, Tallahassee, FL), limiting the growth of vegetative cover in the burn blocks. The extended lack of herbaceous cover may have increased mortality risk, especially on large scale areas. In 2005, we observed similar mortality risk among treatments, associated with above normal rainfall (Nation Weather Service, Tallahassee, FL) and higher cotton rat abundance (Hannon 2006).

Prior studies have documented little or no ef-

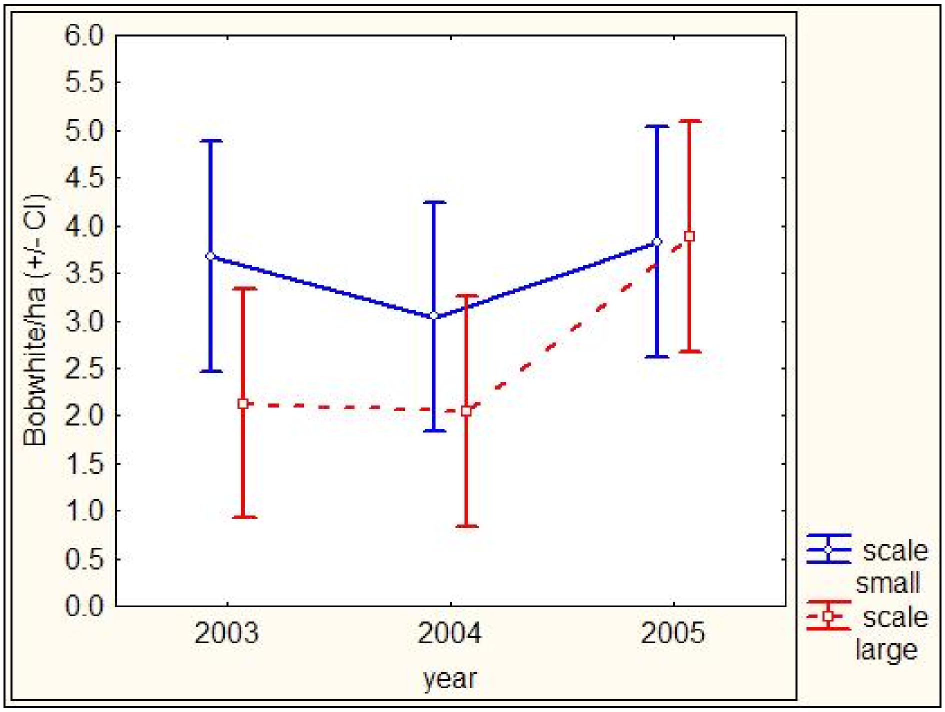

Figure 2: Autumn density of northern bobwhite by small and large scale treatment areas on Tall Timbers Research Station, Florida, USA, 2003-2005.

fect of burning on bobwhite survival. Mueller et al. (1988) documented no differences in mortality rates between clean burns and patchy burns (>100 ha) among years. Carter et al. (2002) observed no differences in survival between bobwhites in burned or unburned pastures (>60 ha). We observed an effect of burning on mortality risk during 2 years in which survival was lower than normal (Palmer and Wellendorf 2007), and we observed autumn population declines those years. Effects of burning scales on mortality risk of bobwhites may be greater during years with generally poor conditions for bobwhites, and during those years burning at small scales may improve survival. However, during relatively good years for bobwhites, such as 2005, scales of burning may not influence survival.

Small scale management may decrease nesting productivity and success due to increased search efficiency of nest predators (Jimenez and Conover 2001, Wiens 1989). Small block sizes allow for more complete searching within a block and increases the amount of edge available for predators to forage along, especially for gray rat snakes (Stapleton 2005) and mammals, such as raccoons, which are both major nest predators (Staller et al. 2005). We observed higher nest success on large scale areas in 2003 and 2005, but documented higher nest productivity for hens on small scale areas all 3 years. The hatch index, which is a combination of the 2 other factors, did not differ between treatments in 2003 and was slightly higher on small scale areas in 2004 and 2005. Most likely, the differences we observed in per capita nest production were due to differences in mortality risk during the early breeding season. The productivity index is greatly influenced by hen survival, especially during the early breeding season, which is when the highest numbers of nests are incubated (Terhune 2004). After accounting for survival, the differences in productivity between scale treatments would likely have been similar which supports the conclusion that burning at different scales would have little impact on productivity of bobwhites.

The productivity index and hatch index yield different conclusions when comparing large scale and small scale areas due to differences in nest success between areas. While the difference in nest success was greatest in 2003, it had little impact on autumn abundance, which supports the idea that nest success is a poor predictor of site productivity (W. Palmer, unpublished data). Additionally, macrohabitat conditions around the nest have little impact on nesting success. Staller et al. (2002) found no differences in macrohabitat composition around successful and unsuccessful nests on TTRS, including both burned and unburned uplands. Their results and the results from this study support the hypothesis that on intensively managed quail lands habitat composition and scale, within the range of this study may have little effect on productivity; other factors such as predator abundance may have a stronger influence on productivity (Staller et al. 2005).

We observed marginally higher autumn densities on small scale treatment areas in 2 of 3 years. The slightly lower mortality risk and improved nest productivity on the small scale areas are likely partial reasons for density differences in the scale treatments, but these factors do not fully explain the varying effect size on the scale treatments by year and other null effects. There is no definitive explanation for the density differences and other factors can affect autumn bobwhite density estimates, such as inherent sampling variability or other unmeasured population parameters including brood survival. Brood survival can have a significant impact on autumn density, and scale of burn may be an important factor, considering the importance of burned upland areas for brood habitat use (Carver 1998, Hammond 2001). Unfortunately, no information is available on brood usage in varying burn scale areas. Movements of bobwhites in and out of our treatment areas between the breeding season and at the time of the autumn covey call surveys could have also influenced our results. We make the assumption that this factor did not impact our results, which reasonable considering the relatively homogeneous habitat conditions and large proportions of usable space in early autumn. Additionally, we did not observe any post-breeding movements from our radio-tagged bobwhites that may have influenced our results.

We observed few differences in home range sizes by scale or year; whereas we had expected to see greater home range size on large scale areas. Hypothetically, upland habitats were to be considered a binomial of either useable or unusable (Guthery 1997) by bobwhites and for a period of time the burned areas would have been considered unusable. Therefore, bobwhites would have been restricted to the unburned habitats after burning and before the early growth phase of the burn blocks, which in turn would have affected home range size. However, we observed almost an immediate use of burned areas and a consistent increase in use over time, more consistent with a gradient concept of increasing habitat suitability over time. Mueller and Atkinson (1985) and Mueller et al. (1988) observed bobwhites allocating their time in burned areas within 7 days after the burn, especially in areas where some residual vegetative cover remained. On TTRS, with a diverse vegetative ground cover associated with post agricultural landscapes, burn blocks typically were a mosaic of various amounts of cover. Additional factors, such as supplemental feeding and predator control may have minimized variability in home range size (Newton 1998), which could have potentially reduced the effects of burning scales on home range sizes.

While small scale treatments had slightly higher productivity and autumn densities, the additional land management costs associated with burning at a small scale may well outweigh the additional population gains. Approximately 4 times more fire breaks were needed in the small scale treatment areas, and time for implementation of the burn was increased to protect the numerous surrounding blocks from burning. The increased disturbance associated with small scale prescribed burning may be unwanted on areas where protecting sensitive native ground cover is a concern.

Table 2: Mean home range sizes for northern bobwhite on small and large scale treatment areas on Tall Timbers Research Station, Florida, USA, 2003-2005.

Year	Scale	*n*	Mean Home Range (ha)	SE
2003	LARGE	29	16.2	1.1
	SMALL	12	13.9	1.3
2004	LARGE	18	17.6	1.7
	SMALL	17	12.6	1.3
2005	LARGE	22	14.2	1.1
	SMALL	23	14.1	1.7

Management Implications

On Tall Timbers Research Station, which implements land management practices similar to many southeastern hunting plantations, a large proportion of the landscape is considered useable for bobwhites throughout most time periods within a year. Prescribed burning of upland areas is the primary land management tool to maintain vegetation characteristics optimal for bobwhites. However, for a short period of time after fire vegetation cover is reduced, bobwhites may be more susceptible to predation and forced to move to suboptimal habitat until cover returns. We attempted in this study to determine if we could improve survival and productivity by adjusting the juxtaposition of burned areas to unburned areas.

Overall, we observed minor differences in survival, productivity, or autumn density at the 2 management scales that we burned. On southeastern hunting plantations, either management scale would be adequate for upland habitat management for bobwhite. We recommend managing burn blocks of a size within the range of this study that is most appropriate for the landscape and other management objectives, such as adjustment of fire intensity and minimizing ground disturbance from establishing fire breaks. During years in which production is relatively poor due to early growing season droughts or excessive avian mortality, small scale burning may have some potential benefits during the early breeding season that could improve autumn density.

While the prescribed burning scales used in this experiment are typical on southeastern hunting plantations they are uncommon on public lands and other properties that manage for early successional habitats critical for bobwhites. Normally, these areas burn at scales significantly larger ranging from 100's to 1000's of hectares in size. Our results indicate that burn scales, which were considerably smaller than those on public areas, can impact bobwhite demographics on some years. Managers of other property types interested in bobwhite management should evaluate burning scales in their resource management plans and burn at the smallest scale whenever possible to improve bobwhite survival and productivity.

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Effect of Supplemental Feeding on Winter Diet and Body Mass of Northern Bobwhites

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Supplemental feeding by spreading grain is a common management practice on private lands managed for northern bobwhites (*Colinus viginianus*), but its effect on diet of bobwhites is not well documented. We examined crop contents from 160 northern bobwhites collected in January and February of 2002 and 2003 from Tall Timbers Research Station, Tallahassee, FL, and Pebble Hill Plantation, Thomasville, GA. Sorghum seed (*Sorghum vulgare*) was spread bi-weekly on approximately half of each study area along a dedicated feed trail through the cover. Dry mass of crop contents from harvested bobwhites averaged 2.51g (SE = 0.171) and was composed primarily of seeds ($\bar{\chi}$ = 2.44 g; SE = 0.171), with minor invertebrate ($\bar{\chi}$ = 0.052g; SE = 0.021) and vegetation ($\bar{\chi}$ = 0.016g; SE = 0.004) matter. Dry mass of crop contents for bobwhites averaged 2.94 g (SE = 0.280) for birds from fed sites and 2.09g (SE = 0.187) from reference sites. On fed sites 67.5% of bobwhites had used sorghum and sorghum accounted 71% of food items by weight. Diet varied annually, primarily with changing use of acorn, (*Quercus* spp.) pine (*Pinus* spp.) mast, and partridge pea (*Cassia* spp.). Adult bobwhites captured in January averaged 169.9g (SE = 0.970; n = 181) on reference sites and 174.8g (SE = 0.896; n = 237) on fed sites. Supplemental feeding by spreading sorghum was an effective method of providing seeds with high metabolizable energy to bobwhites.

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Key words: *Colinus virginianus*, Florida, Georgia, milo, northern bobwhite, quail, *Sorghum vulgare*, supplemental feed

Introduction

Food supplementation has a long history in bobwhite management (Frye 1954) and supplemental feeding has since replaced food plot planting on many private lands managed for bobwhites. Unlike food plot plantings, supplemental feeding is not dependent on soil or weather conditions, can be made available to the bobwhites year-round, provides food for many wildlife species, and is applicable where disking ground or using agrichemicals to plant crops is not possible (e.g., erosion potential or wet conditions) nor acceptable (e.g., disrupting sensitive or threatened plant communities). For these reasons and others (Sisson et al. 2000), landowners managing for bobwhites are adopting food supplementation.

A few studies have investigated the use of sorghum by bobwhites and the effects of supplemental feeding on bobwhite demographics. In Georgia, Michael and Beckwith (1955) showed that sorghum was the most preferred food of native and planted species tested. Robel and Arruda (1986) reported that bobwhites maintained body weight on sorghum, but not some natural foods prevalent in winter diets. In Kansas, Robel (1969), Robel et al. (1974) studied bobwhite use of food plots and reported those near food plots weighed more and contained a higher percentage of body fat than birds without food plot available to them. Frye (1954) demonstrated population increases on a portion of a wildlife management supplementally fed using feeders. Sisson et al. (2000) reported higher winter survival rates and smaller home ranges when bobwhites were provided supplemental feed through spreading in high quality habitat. Finally, Townsend et al. (1999) reported bobwhites in Oklahoma used

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sorghum from feeders even when food plots were available, exhibited higher survival in 2 of 4 years of their study, and were more resistant to severe weather events. However, Guthery (1999) found little support for the food-limitation hypothesis in quail management.

Few studies have compared the relative importance of supplemental feed in the diet of bobwhites relative to control bobwhites. Frye (1954) found heavy use of supplemental feed from feeders during a year with low natural foods in Florida. Robel et al. (1974) reported sorghum was 27.9% of crop contents for bobwhites near food plots. Townsend et al. (1999) reported percent sorghum of bobwhites harvested during winter from an area with supplemental feeders was 28.2% relative to 5.5% for bobwhites from a control area. No studies have reported the diet of bobwhites on areas supplementally fed by spreading feed, which is becoming a common practice on managed properties in the southeast. Therefore, we determined the effect of supplemental feeding on diet and body mass of bobwhites on 2 areas during late winter.

Study Area

We studied bobwhites during January and February on Tall Timbers Research Station (1,568 ha) (TTRS), Leon County, Florida and Pebble Hill Plantation (1,246 ha) (PH), Thomas and Grady Counties, Georgia. These study areas, which are 12 km apart, were located in the Red Hills physiographic region in the Gulf Coastal Plain of northern Florida and southern Georgia. Our study areas consisted of upland pine forests (80%) dominated by loblolly pine (*P. taeda*) and shortleaf pine (*P. echinata*) with associated "old-field" ground cover vegetation on TTRS. On PH, approximately 30% of the study area is longleaf pine (*P. palustris*) with associated wiregrass (*Aristida stricta*) ground cover. Intermixed throughout the study sites were hardwood drains and hammocks and annually disked fallow fields. Basal area of overstory trees in upland areas managed for bobwhites averaged 9.2 m^2/ha on TTRS and 10.5 m^2/ha on PH. Approximately 60-70% of each site was burned annually to promote herbaceous ground cover and reduce the stature of hardwood shrubs and saplings.

Methods

Supplemental Feeding

The northern portion of TTRS (445 ha) and the southern portion of PH (532 ha) were supplied with sorghum once every 2 weeks on a year-round basis. Sorghum was spread from a tractor along a specified route through the uplands. The feeding route covered 19 km on TTRS and 14 km on PH. Difference in length of feeding routes between study areas resulted from differences in landscape characteristics between the study sites. Pebble Hill has more hardwood drains than TTRS which limited tractor access to some areas. Sorghum was spread continuously along the route at the rate of approximately 95 kg of sorghum per km of feeding route. For example, this amounts to 107 kg of sorghum per ha per year (1.7 bushels of sorghum/acre/year) for the TTRS study area. Sorghum was used because it is preferred by bobwhites (Michael and Beckwith 1955) and has excellent energetic properties for bobwhites (Robel and Arruda 1986).

Data Collection

We collected 160 hunter-shot bobwhites during January - February, 2002 - 2003. Forty bobwhites were collected from the fed area and reference area of TTRS and PH each year of the study. Each bobwhite was aged and weighed to the nearest 0.1 g. Crops were removed from bobwhites and placed individually in plastic bags and marked by study area, date, age, sex, and band history. Crops were frozen until further analyses.

Crops were thawed then their contents emptied into a #25 sieve. Contents were washed to separate vegetation, insects and seeds. We placed all of the vegetation in a dish for drying. We then removed the insects and placed them in a tube containing a 70% ethanol solution. We transferred the remaining contents, which were now composed only of seeds, to a dish and oven-dried seeds and vegetation at 80°

C for a period of 24 hours.

We separated seeds from each crop by genus, according to Landers and Johnson (1976), using a magnifying lamp and a dissection microscope. Once all of the seeds were separated, we counted and weighed the dried seeds. We also dried and weighed the insect and vegetation matter collected. Using this data, we calculated mean crop weight, mean seed weight, mean % of seed per crop, and mean number of seeds per crop.

We weighed captured bobwhites to determine their body mass on TTRS and PH during January in baited-funnel traps. Details on hunting procedures and capturing procedures are published elsewhere (Palmer et al. 2002). Bobwhites were weighed to the nearest 0.1 g on an electric balance. Bobwhites on both fed and reference sites were captured in traps baited with sorghum, therefore, mass estimates are biased high, but the comparison among fed and reference sites should not be biased.

Results

Crop Contents

Dry weight of crop contents averaged 2.51g (SE = 0.171) and was composed primarily of seeds ($\bar{x}$ = 2.44 g; SE = 0.171), with lesser amounts of invertebrate ($\bar{x}$ = 0.052 g; SE = 0.021) and vegetation ($\bar{x}$ = 0.016 g; SE = 0.004) matter. Dry weight of crop contents for bobwhites from fed sites was 2.94 g (SE = 0.280), whereas mean mass of crop contents from bobwhites from reference sites was 2.09 g (SE = 0.187). Bobwhites from fed sites ate slightly less vegetation ($\bar{x}$ = 0.0064 g; SE = 0.0032) than bobwhites from reference sites ($\bar{x}$ = 0.0261 g; SE = 0.0073).

Mass of crop contents also varied by year, averaging 2.66 g (SE = 0.236) per crop in 2003 and 2.36 g (SE = 0.248) per crop in 2002. Mass of crop contents varied more among years for bobwhites from reference sites than fed sites. Bobwhites on reference sites averaged 1.82 g (SE = 0.274) in 2002 and 2.35 g (SE = 0.251) in 2003. Whereas, mean mass of crop contents for bobwhites on fed sites was 2.90 (SE = 0.399) in 2002 and 2.98 (SE = 0.397) in 2003.

Relative Use Of Sorghum

We found 32 species of seeds in the 180 quail crops analyzed. There were 8 species of seeds found from bobwhites in reference sites that were not present in bobwhites from fed sites (Table 1). Of bobwhites with some crop contents, 67.5% of bobwhites harvested on fed areas had sorghum in their crops which accounted for 65.9% (95% CI = 56.2 - 75.7) of the crop contents mass. Including only bobwhites that had sorghum in their crops, sorghum comprised 91.5% of the crop contents, averaging 2.94 g (SE = 0.351) per crop. The most sorghum found was in a single crop was 9.8 g in a juvenile female, which also happened to be the heaviest crop contents of the 80 bobwhites harvested from fed sites.

Annual Variation

Bobwhites used different foods each year. Mass of acorn meat in crops was 16.7x greater in 2002 ($\bar{x}$ = 0.50 g, 95% CI = 0.243 - 0.749) than 2003 ($\bar{x}$ = 0.03 g, 95% CI = -0.035 - 0.103). Pine mast also was more prevalent in crops in 2002 ($\bar{x}$ = 0.14 g, 95% CI = 0.045 - 0.024) than 2003 ($\bar{x}$ = 0.05 g, 95% CI = 0.024 - 0.071). In 2002, when pine and oak mast use was relatively high, bobwhites on fed sites utilized these foods ($\bar{x}$ = 0.32 g, 95% CI = 0.079 - 0.560), but less so than on reference sites ($\bar{x}$ = 0.99g, 95% CI = 0.481 - 1.497). In 2003, partridge pea was the most prevalent native seed in crops of bobwhites ($\bar{x}$ = 1.18g, 95% CI = 0.828 - 1.53), and it was used significantly less in 2002 ($\bar{x}$ = 0.17 g, 95% CI = 0.061 - 0.263). In 2003, partridge pea was utilized more on reference areas ($\bar{x}$ = 1.62g, 95% CI = 1.17 - 2.07) than fed areas ($\bar{x}$ = 0.70g, 95% CI = 0.183 - 1.210). One bobwhite had 1287 partridge pea seeds in its crop. Mass of all other legumes combined did not vary from reference ($\bar{x}$ = 36g, 95% CI = 0.199 - 0.512) and fed sites ($\bar{x}$ = 0.33 g, 95% CI = 0.088 - 0.568). All other seed types combined accounted for an average of 0.075 g (SE = 0.023) per crop.

Body Mass

Adult bobwhites captured in January averaged 169.9 g (SE = 0.970; n = 181) on reference sites and 174.8 g (SE = 0.896; n = 237) on fed sites and the 95% CI's did not overlap. Similarly, captured juvenile

Table 1: Percent use and dry mass (g) of seeds in crops (when present) of northern bobwhites (n = 80) collected from sites with sorghum (*S. vulgare*) spread along dedicated feeding trails and reference sites (n = 80) on Tall Timbers Research Station, Florida, and Pebble Hill Plantation, Georgia, in January and February, 2002–2003.

	Supplementally-fed						Reference				
Seed type	n	Use %	Dry mass	Min	Max		n	Use (%)	Dry mass	Min	Max
Sorghum vulgare	54	67.5	2.939	0.037	9.768		9	11.3	1.579	0.017	3.476
Desmodium spp.	32	40	0.092	0.002	1.425		37	46.3	0.143	0.003	1.294
Lespedeza spp.	23	28.7	0.281	0.003	5.668		39	48.8	0.181	0.002	4.439
Chamaecrista spp.	35	43.8	0.894	0.004	6.155		49	61.3	1.451	0.008	4.915
Pinus spp.	30	37.5	0.129	0.004	1.156		41	51.2	0.251	0.006	3.204
Galactia volubilus	23	28.7	0.63	0.006	5.694		37	46.3	0.212	0.006	1.577
Acorn (*Quercus* spp.)	6	7.5	1.491	0.039	2.856		19	23.8	1.577	0.043	5.515
Zea mays	3	3.8	0.302	0.14	0.494		2	2.5	2.039	1.375	2.703
Geranium spp.	0	-	-	-	-		2	2.5	0.005	0.005	0.006
Centrosema virgininanum	12	15	0.059	0.011	0.168		36	45	0.175	0.001	1.35
Panicum spp.	9	11.3	0.024	0.002	0.141		9	11.3	0.007	0.001	0.017
Tephrosia virginica	1	1.3	0.01	0.01	0.01		6	7.5	0.031	0.004	0.067
Trichostema dichotomum	12	15	0.052	0.002	0.366		25	31.3	0.042	0.002	0.206
Juncus spp.	9	11.3	0.01	0.005	0.018		15	18.8	0.017	0.002	0.06
Ambrosia artemisiifolia	4	5	0.021	0.007	0.039		6	7.5	0.129	0.003	0.733
Helianthus spp.	1	1.3	0.018	0.018	0.018		5	6.3	0.018	0.006	0.052
Acalypha spp.	5	6.3	0.004	0	0.008		12	15	0.005	0.001	0.021
Cercis canadensis	0	-	-	-	-		1	1.3	0.025	0.025	0.025
Liquidambar styraficula	0	-	-	-	-		4	5	0.14	0.007	0.312
Polygonum spp.	0	-	-	-	-		1	1.3	0.001	0.001	0.001
Rumex acetosella	0	-	-	-	-		1	1.3	0.003	0.003	0.003
Parthenocissus quinquefolia	0	-	-	-	-	1	1.3	0.034	0.034	0.034	
Prunus serotina	0	-	-	-	-		2	2.5	0.66	0.053	1.267
Richardia scabra	1	1.3	0.062	0.062	0.062		4	5	0.047	0.002	0.167
Diodia teres	1	1.3	0.007	0.007	0.007		1	1.3	0.015	0.015	0.015
Bouteloua curtipendula	0	-	-	-	-		1	1.3	0.012	0.012	0.012
Unknown	4	5	0.007	0.002	0.016	7		8.8	0.028	0.001	0.161

bobwhites averaged 167.3 g (SE = 0.486; n = 820) on reference sites and 171.5 g (SE = 0.464; n = 1004) on fed sites and these 95% CI's also did not overlap. The mean difference in body mass between fed and reference sites averaged 4.3 and 4.8 g for adult and juvenile bobwhites, respectively. Body mass was consistently lower on reference sites each year at TTRS, however, on PH, body mass was lower in 2003 than in 2002 on the reference site (Figure 1).

Discussion

Bobwhites on our study areas used sorghum greater than any other seed when it was available. This was not surprising given that the supplementation program was designed make sorghum available to most bobwhites on our study areas and sorghum is a highly preferred bobwhite food (Michael and Beckwith 1955). Studies reporting use of food plots and feeders reported lower use than our study. Robel et al. (1974) reported that 27.9% crop contents was sorghum for bobwhites living near food plots. Townsend et al. (1999) reported sorghum from feeders composed 28.2% (Range 20.7 - 45.2%) of crop contents from bobwhites harvested during winter. These compare to 71% of crop contents, by weight, in this study. The relatively high use of sorghum in our study suggests that supplemental feeding by spreading along dedicated trails is more effective at provided sorghum to bobwhites than feeders or food plots.

Diets of bobwhites were diverse on both the fed and reference areas, although some minor diet items were found only on the reference areas. The most important native foods included partridge pea, acorn mast, pine mast, lespedeza, milk pea (*Galactia volubilis*) and beggar-tick (*Desmodium* spp). Therefore, while bobwhites made heavy use of sorghum seeds, this did not result in complete reliance on supplemental feed. Similar results were found by Robel et al. (1974) and Townsend et al. (1999).

Robel et al. (1974) compared the metabolizable energy (ME) per crop for bobwhites near and far from food plots and reported ME of 7514 cal/crop for birds utilizing food plots and 5653 for bobwhites not using food plots. We developed an estimate of ME per crop for bobwhites in this study using energy content and ME values provided by Robel et al. (1974, pg. 660). For species lacking data on ME, we used the reported overall mean ME for the species they tested. For bobwhites on the fed portion of the study area, the mean crop contained roughly 9720 cal versus 5531 cal for bobwhites on the reference sites. The ME per crop for reference bobwhites in this study was similar to that reported by Robel et al. (1974) for bobwhites not utilizing food plots. However, ME per crop for bobwhites on the fed portion of our study areas was $\sim$2500 cal higher than those utilizing food plots in Robel et al.'s study due to double the sorghum content in crops from this study. On fed sites in this study, 76% of the energy per crop came from sorghum versus 42% in Robel et al.'s study. Collectively these results suggest that supplemental feeding through spreading is more effective at providing high energy foods to a bobwhite population than food plots.

Some insights may be gained from evaluating ME of bobwhites with full crops, as most bobwhites were harvested during feeding activity. For instance, a juvenile female had 385 sorghum seeds in her crop which is approximately 36,280 cal. This amount of sorghum indicates bobwhites could feed 1 to 2 times per day to meet daily ME needs at 2° C (Robel et al. 1979). This compares to 1287 partridge pea seeds in the crop of a juvenile male on the unfed site, approximately 21,420 cal. Differences in diet and energy content suggest total feeding time may be as much as 3 to 4 times greater on the unfed sites than fed sites. Sisson et al. (2000) reported smaller home ranges and higher survival for bobwhites with access to supplemental feed during some winters and suggested that observed lower mortality was an artifact of decreased foraging activity and predation risk. Similarly, Townsend et al. (1999) suggested supplemental feeding may reduce exposure of bobwhites to predation. Home ranges on our study areas were 50% smaller on fed sites than unfed sites and Kaplan-Meier winter survival was significantly higher on fed sites in some years (Palmer and

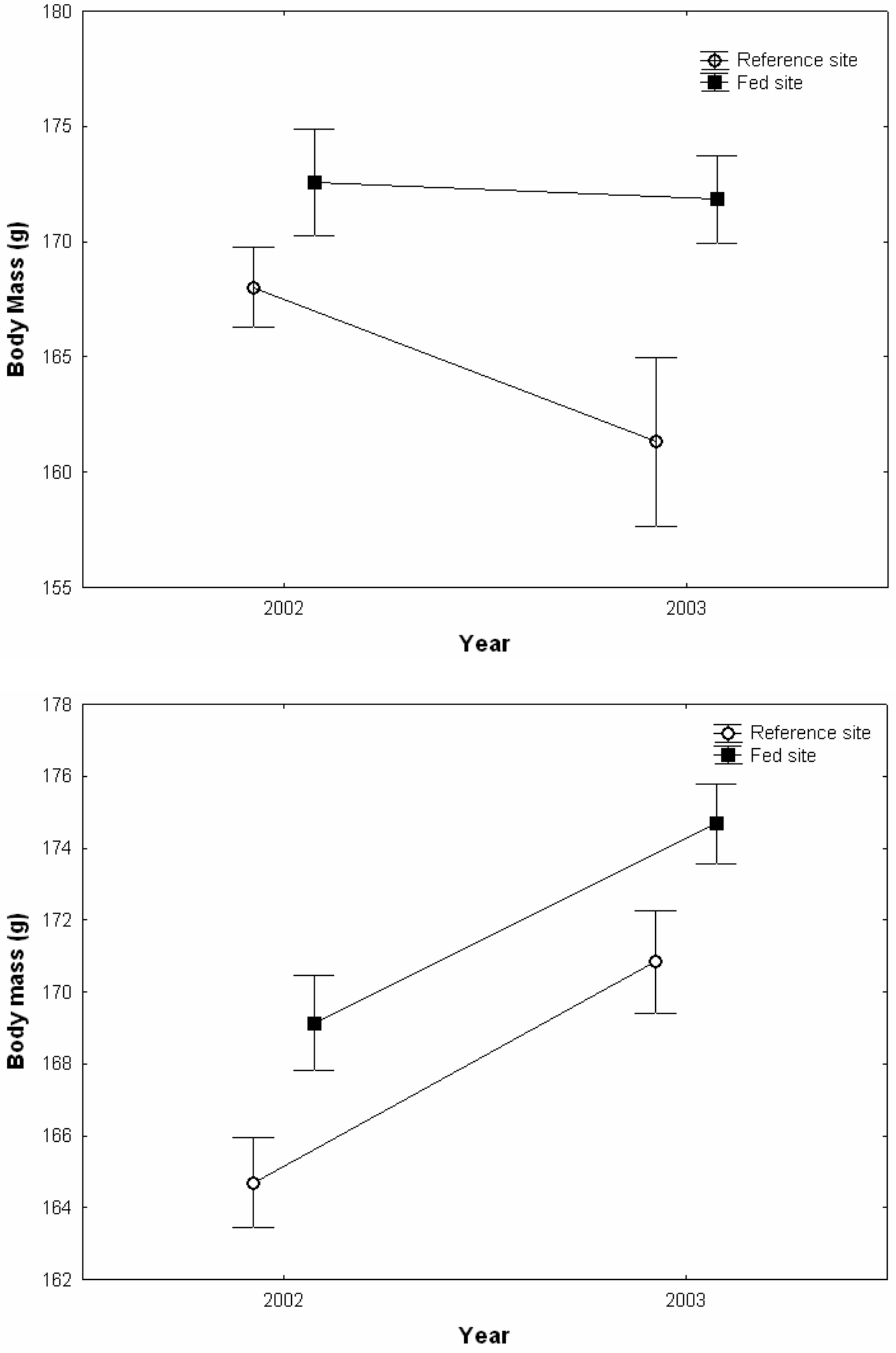

Figure 1: Average mass (g) and associated 95% confidence intervals for northern bobwhites (n = 852) captured on fed and reference areas on Pebble Hill (top graph), Grady County, Georgia and Tall Timbers Research Station (lower graph), Leon County Florida, 2002–2003.

Wellendorf, unpublished data). Robel and Kemp (1997) suggested higher survival for birds with access to food plots was due to improved resistance to severe winter weather. Guthery (1999) countered that increased lipid loads in bobwhites with access to food plots would provide marginal improvement in resistance to periods of severe weather. In the deep south, where winter weather is almost never severe (e.g., extended periods of snow cover) and natural food supplies are adequate in good habitat (Guthery 1999), we hypothesize that supplemental feeding may lessen predation on bobwhites by reducing foraging time and/or modifying foraging behaviors. Future studies of supplemental feeding should consider the effect of foraging time and foraging behavior on bobwhite survival.

Greater body mass of bobwhites on fed areas in this study was consistent with other studies (Robel 1969, Robel et al. 1974) although the difference was lower than that reported by Robel et al. (1974) in Kansas. In addition to sustaining higher body weight, body weight varied little each year on supplementally-fed sites. In contrast weights varied annually on reference sites and were relatively low in 2003 on the reference portion of PH. We suspect low acorn and pine mast availability during 2003 resulted increased use of partridge pea in 2003. On reference sites in 2003, 71% of the dry mass of crop contents for bobwhites with >1 g of seeds in their crops (n = 30) was partridge pea. This contrasts with 7% of crop contents for birds similarly sampled in 2002. Partridge pea is a relatively poor food energy source for bobwhites because of its low metabolizable efficiency (Robel et al. 1974). Robel et al. (1974) also reported relatively heavy use of low energy foods during January and February, including sumac and partridge pea. Bobwhites may adjust for lower energy foods by increasing volume ingested (Giuliano et al. 1996). In our study bobwhites on reference sites had slightly heavier crops in 2003 than 2002 suggesting increased weights were a result of reliance on seeds with lower energy content.

Management Implications

Supplemental feeding of bobwhites by spreading sorghum seed along feeding routes through properly managed habitat reduced annual variation in diet composition and ME and resulted in increased body mass during late winter. Our results suggest that supplemental feeding reduces reliance on poor quality foods (e.g., low ME) in late winter, even in an area in the "deep south" that had a diverse plant community and abundant native foods that support bobwhites (Stoddard 1931).

Providing high energy foods for bobwhites likely reduces foraging time to meet daily energy needs. In this study, bobwhites on fed sites had roughly 50% more energy per crop suggesting feeding time was reduced by at least that amount. Other studies have suggested that providing high energy foods through supplemental feeding (Townsend et al. 1999, Sisson et al. 2000) or food plots (Robel and Kemp 1997) resulted in higher over winter survival rates in at least some years. Reducing over-winter mortality is an important management objective for sustaining bobwhite populations.

Supplemental feeding as a food management technique is preferred over food plantings because it does not depend on suitable growing conditions to produce, can be adapted quickly to changing conditions (flooding, habitat changes etc.) and does not result in soil disturbance nor require the inputs such as fertilizers and pesticides, and is more effective at delivering high energy foods to bobwhites. While we did not directly measure the costs associated with supplemental feeding, we suspect it is not outside the budget of many wildlife management programs on public or private lands. Further research is needed to determine if population level responses accrue from changes to demographic parameters as a result of supplemental feeding in areas of good to excellent habitat.

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Policy and Conservation

The GWCT Grey Partridge Recovery Programme: *a Species Action Plan* in Action

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In 1994, the UK government launched its Biodiversity Action Plan (BAP). Its list of BAP species included the grey partridge *Perdix perdix*, whose UK numbers had declined by 91% since 1962. The government set numerical targets in its grey partridge Species Action Plan, then nominated the Game & Wildlife Conservation Trust (GWCT) as lead partner to take the Plan forward. To this end, the GWCT has sought to raise awareness of the issues among the farming and shooting communities with promotional material. It has encouraged land and shoot managers to join its Partridge Count Scheme and established local Partridge Groups as a focus for information dissemination, guiding management with local targets set using landscape characteristics to estimate the potential distribution of grey partridges at the 1-km^2 level. Leading by example, the GWCT has demonstrated on the ground how appropriate management leads to successful grey partridge recovery, and it has initiated research into optimal methods of re-establishing grey partridges in areas of extinction. I consider the current population status of the grey partridge in the UK, review the targets of the Species Action Plan in the light of that status, and discuss the chances of success.

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Key words: action plan, grey partridge, *Perdix perdix*, population recovery, UK

Introduction

In June 1992, the UK Prime Minister and heads of state from over 150 countries signed the Convention on Biological Diversity at the Earth Summit in Rio de Janeiro, Brazil. Two years later, the UK government published its first Biodiversity Action Plan (BAP), which sought to develop a framework to conserve biodiversity in the UK (Anonymous 1994). The Steering Group set up to implement it identified priority species and habitats in need of conservation and published action plans for them (Anonymous 1995).

Prominent among the species was the grey partridge *Perdix perdix*, whose numbers in Britain had declined by 88% between 1962 and 1988 (Marchant et al. 1990), and whose range had contracted by 19% between 1970 and 1990 (Gibbons et al. 1993). The grey partridge Species Action Plan (Anonymous 1995) defined three targets for population restoration: halt the decline by 2005, ensure that the population is above 150,000 pairs by 2010, maintain and where possible enhance the current range of this species. The yardsticks against which to judge performance against these particular targets were the bird surveys organised by the British Trust for Ornithology (BTO): the Breeding Bird Survey (formerly the Common Birds Census) for a national index of abundance since 1962 (see Figure 1), involving over 2000 random 1-km^2 squares visited annually (Newson et al. 2005), and the Breeding Bird Atlas surveys for the assessment of range, involving complete coverage of all 100-km^2 squares in the UK every 20 years since 1968 (Sharrock 1976, Gibbons et al. 1993).

In 1996, the UK government nominated the Game & Wildlife Conservation Trust (GWCT) as lead partner responsible for implementing the action plan for grey partridge. No government funding accompanied the nomination, but thanks to the generosity of private individuals and companies, the GWCT raised enough money to launch a major programme for partridge recovery. Because almost all UK land is privately owned, and land ownership

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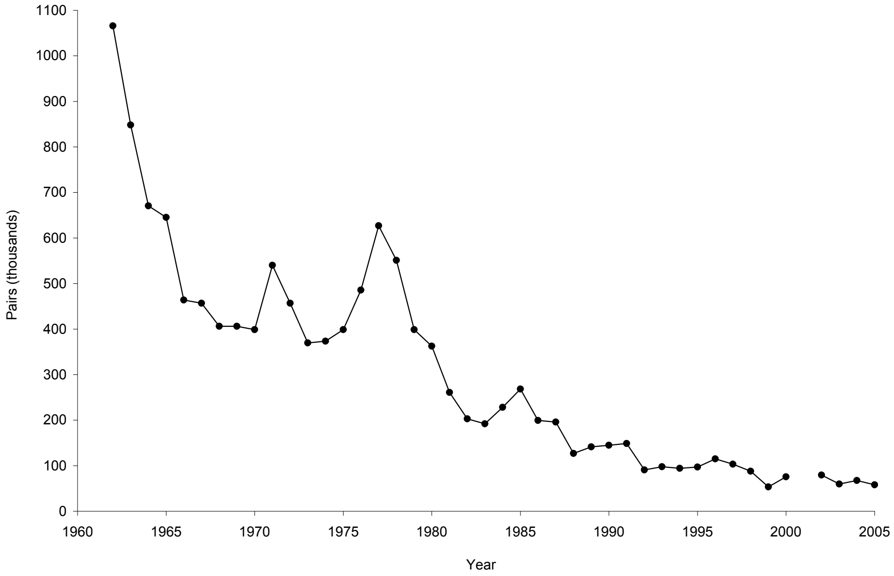


Figure 1: Grey partridge abundance in the UK, 1962-2005, based on annual indices from British Trust for Ornithology surveys (Marchant et al. 1980 and Newson et al. 2005, updated), calibrated by reference to the 1988-91 Atlas of Breeding Birds (Gibbons et al. 1993).

confers ownership of any game present, the cornerstone of the programme was to motivate farmers, land owners and shoot managers to conserve grey partridges on their land.

The programme was primarily education-oriented rather than research-based because, already in the 1970s, GWCT research had identified agricultural intensification and increased predation losses as the principal drivers of partridge decline (Potts 1980, 1986). In brief, nesting cover disappeared following hedgerow removal to improve farming efficiency, brood production fell because of increased pressure from avian and mammalian predators, and chick survival dropped when pesticide use depleted numbers of chick-food insects in cereals. During the 1980s and 1990s, the GWCT developed crop and margin management techniques to mitigate these effects without compromising farming profitability (Aebischer 1997). These include selectively sprayed cereal margins (”conservation headlands”) to restore beneficial cereal insects, and raised mid-field strips of tussocky grass (”beetle banks”) to provide nesting cover and biological pest control. Turning to advantage the European Union’s farm subsidy requirement of setting aside a percentage (8% in 2006) of arable land from crop production, they also include growing unharvestable crop mixtures (e.g. mustard and triticale, or kale and quinoa) on set-aside land to provide nesting, brood-rearing and overwinter cover, ideally in combination.

GWCT Grey Partridge Recovery Programme

The GWCT recovery programme seeks to influence management at the farm level in several different ways, as reviewed below.

Educational Material

The GWCT has put together a range of leaflets to raise awareness of the grey partridge conservation needs, all of which are available via the internet (http://www.gwct.org.uk/partridge).

The main summary leaflet ”Conserving the Grey Partridge” is eight pages long. It was endorsed by the Farmland Birds Species Action Plan Steering Group, comprising representatives of government, statutory conservation bodies and NGOs, and is aimed at the general public as well as practitioners and policy-makers. It describes the BAP context, the status of the grey partridge and general management measures to improve land for grey partridges (during the nesting period, when suitable nesting cover is required, the chick-rearing period, when insect availability is of paramount importance early in life, and the overwinter period, when food and shelter are often lacking in the modern agricultural landscape). The leaflet also provides guidelines on when to shoot or not to shoot grey partridges, emphasizing the need to take precautionary measures during driven shooting based on released red-legged partridges (Aebischer and Ewald 2004).

A series of six fact sheets address management issues in greater detail for farmers who would like to help the grey partridge on their land. The sheets cover specific habitat requirements (”Restoring grey partridges to your farm”, ”Providing nesting cover for wild grey partridges”, ”Providing brood-rearing cover for wild grey partridges”, ”Providing winter cover and food for wild grey partridges”), the methods of controlling predators (”Using predation control to increase wild grey partridge numbers”) and how to make use of government agri-environment subsidies to best effect (”Environmental Stewardship: making the most for grey partridges”).

Two further leaflets advertise the GWCT’s Partridge Count Scheme and its demonstration project at Royston (see below).

Partridge Count Scheme

The GWCT’s Partridge Count Scheme (PCS) began in 1933. Originally, it was a means of monitoring annual densities and breeding success through a network of around 90 gamekeeper participants who counted the birds on their land in spring and autumn. The GWCT relaunched the scheme in 1998 under the banner ”Every one counts”. It sought to increase the national coverage and, in addition to the monitoring role, to use the contact with gamekeepers, farmers and landowners to encourage more and better management. Core to the approach was persuading such people that, even if they had few grey partridges on their land, it was worth making the effort to conserve them because every increase contributed to the restoration of the species. To help contributors count and monitor their own partridges, the GWCT produced a guide to aging and sexing grey partridges in the spring and autumn. In addition, each contributor receives a spring and autumn newsletter, a minimum numbers of pairs to aim for (based on landscape characteristics - see below) and feedback on how to achieve this. The paper by Ewald et al. (2009) goes into the PCS in greater depth.

Local Partridge Groups

Bringing together people who are interested in grey partridges promotes enthusiasm and opportunities for information dissemination. In counties (or clusters of counties) with over 30 PCS contributors, the GWCT organises local Partridge Groups with at least two meetings a year open to all contributors within the area. The meetings allow presentation and discussion of the latest research, management ideas and government agri-environment regulations relevant to grey partridges, and also comprise field visits showing good management practice. In 2006 there are nine Partridge Groups, in the Borders, Cotswolds, East Lothian, Kent/Sussex, Lincolnshire, Norfolk, Northumberland, Shropshire/Cheshire/Staffordshire and Wessex, with more planned.

Restoration Project

The GWCT’s demonstration farm at Loddington has shown spectacularly for pheasants and hares how much may be achieved for relatively modest effort, and welcomes over 2000 visitors a year (Stoate and Leake 2002). The same type of demonstration was sorely needed for grey partridges. In 2001, the GWCT therefore set up a new demonstration site where visitors may see for themselves the management techniques that are needed for grey partridges, observe the increase in numbers of grey partridges that results from the management, and learn about the pitfalls and costs. This restoration project is so important that it deserves a section all of its own (see below).

Research into Re-establishment

The approaches described above attempt to address the first of the grey partridge Species Action Plan targets, lay the foundations for the second, but fail to tackle the third. The problem is that the grey partridge has now disappeared from large parts of its former range, and is at very low density in others. By 2003, anecdotal evidence was mounting that, despite appropriate management, grey partridge restoration was unsuccessful in some areas because of a lack of wild birds to take advantage of it. Re-establishment through releasing was the obvious solution, especially as the grey partridge is relatively easy to rear in captivity. However, past work has shown that the standard practice of releasing young game-farm birds in late summer fails repeatedly because of poor post-release survival. In 2005, the GWCT therefore launched a new research project "Determining optimal release methods as a tool for restoring the abundance and range of grey partridges in the UK". Browne et al. (2009) review releasing techniques and describe the experimental design of the research trial.

Estimating Partridge Distribution

Underpinning the whole of the GWCT's grey partridge recovery programme is the ability to identify what parts of the UK are suitable for grey partridges, and to assess their potential in terms of the likely density of breeding pairs - it would be self-defeating to encourage species recovery in intrinsically unsuitable locations, or to raise unrealistic expectations.

We built on Tapper (1999), who constructed national habitat maps for game and predator species using the Countryside Information System (Department of the Environment, Transport and the Regions and Centre for Ecology and Hydrology), a mapped repository of landscape, land use and climatic data defined for every 1-km^2 grid square (100 ha) across Great Britain. Most data came from the Countryside Survey 1990 and the Land Cover Map of Great Britain (Barr et al. 1993), which provided a national snap-shot of the British countryside from 1988 to 1990 involving detailed field observations and satellite imagery. Ten years later, the Countryside Survey 2000 and Land Cover Map 2000 provided a second snapshot of land cover from 1998 to 1999 (Firbank et al. 2003). Within the Countryside Information System, the Ordnance Survey's 1:250,000 "Strategi" digital map data contributed information on roads and urbanisation. We imported the land cover data into the geographical information system Mapinfo Professional 8.0 (Mapinfo Corp., Troy, New York).

Following Tapper (1999), we first excluded squares influenced by urban areas, specifically ones that met any of the following criteria: >1 ha of town, >25 ha of village, >2 ha of motorway or >6 ha of A-roads. Out of the remainder, squares containing potentially suitable grey partridge habitat were those with >10 ha of arable/horticultural land and <10 ha of deciduous/coniferous woodland. These were further classified into optimal (≥50 ha of arable/horticultural land) and suboptimal (the balance). These definitions gave a reasonable approximation to the open arable habitat where the bird was common in the past, as well as taking in the fringe upland habitat on the edge of cultivated ground that also supports this species. The resulting distribution of squares (Figure 2) closely resembled the observed distribution of the grey partridge in the 1968-1972 Atlas (Sharrock 1976), before the partridge decline began in earnest.

To translate the habitat map into potential numbers of birds, we multiplied the area of each habitat type by the potential density expected there: under modern agriculture, around four, two and zero pairs per km^2 on optimal, suboptimal and unsuitable ground respectively (Potts 1986, Aebischer 1991). The resulting estimates yield UK totals of 219,360 pairs based on the 1990 survey, and 206,160 pairs from the 2000 one (Table 1). The difference between surveys is due to a 30% loss of suboptimal

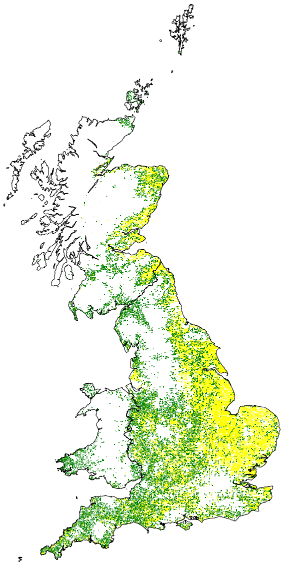


Figure 2: Map of Great Britain showing the national distribution of 1-km^2 squares classified as optimal (yellow), suboptimal (green) and unsuitable (white) for grey partridges based on landscape characteristics in 2000 (see text for details).

ground, offset to some extent by a 7% gain in optimal ground. The changes resulted mainly from the reclassification of suboptimal ground: a fifth of suboptimal squares became optimal following increases in arable, while over a third became unsuitable, principally through the expansion of woodland.

Despite the changes on the ground during the 1990s, the potential number of partridge pairs remains well over the 2010 target figure from the grey partridge Species Action Plan. This is reassuring, because it means that recovery is not an impossible task.

Based on simulation modelling (Potts 1986, Aebischer 1991), the effects of management are roughly to double the density in the case of either predation control or habitat management implemented separately, or to multiply it sixfold (synergistic effect) when both are implemented together. Projecting backward to the 1950s, before the intensification of agriculture and the widespread cessation of gamekeepering, most of Britain would qualify as managed in this way. Assuming a land classification roughly the same as in Table 1, the partridge potential was for 1.2-1.3 million pairs. This fits remarkably well with Figure 1, where an estimated 1.1 million pairs existed in 1962, when the decline had just started.

Grey Partridge Demonstration Project

The Grey Partridge Demonstration Project began in 2002, with as specific aims (1) to develop an area of arable farmland as a demonstration site to restore the abundance of wild grey partridges to a level predicted by GWCT models, and (2) to demonstrate how to manage farmland to increase densities of wild grey partridges in accordance with, and for the furtherance of, the targets laid down in the grey partridge Species Action Plan.

The demonstration area covers 996 ha comprising six farm holdings on light chalky land near Royston, Hertfordshire, in eastern England (Figure 3). A surrounding area of 1311 ha (seven holdings) constitutes a reference area for comparison. Using the mapping approach described above, the amount of optimal and suboptimal land was 723 and 99 ha respectively on the demonstration area, and 1161 and 30 ha on the reference area. In the absence of man-

Table 1: Areas of optimal, suboptimal and unsuitable habitat for grey partridges in England, Scotland and Wales, derived from Land Cover Maps (LCM) for 1990 and 2000, and estimated potential number of pairs (see text for method of estimation).

	Area (km^2)			
Country	Total	Optimal	Suboptimal	Grey partridge potential (pairs)
LCM 1990				
England	130,383	31,884	28,766	185,068
Scotland	78,479	3,802	7,556	30,320
Wales	20,757	30	1,926	3,972
Total	229,619	35,716	38,248	219,360
LCM 2000				
England	130,383	33,666	20,047	174,758
Scotland	78,479	4,497	4,678	27,344
Wales	20,757	77	1,875	4,058
Total	229,619	38,240	26,600	206,160

agement, this gives a potential number of pairs on the demonstration area of 31 (3.1 per km^2), and on the reference area of 47 (3.6 per km^2). The target with full predator and habitat management is 186 pairs (18.6 per km^2) on the demonstration area.

Since January 2002, the GWCT has taken the following measures to increase wild grey partridge densities, based on its understanding of grey partridge ecology:

(a) Predation control. The GWCT employs a gamekeeper whose main duty is the legal control of predators that kill adult partridges or destroy their nests.

(b) Habitat management. The GWCT has encouraged farmers to undertake management that increases the amount of nesting, brood-rearing and overwinter cover, making best use of set-aside and, where possible, linking in with existing agri-environment options subsidised by government.

(c) Supplementary feeding. A secondary duty of the gamekeeper is to provide wheat grain in hoppers placed along field margins and cover strips from September to March, to counteract any winter food shortage.

(d) Other game species. Wild pheasants *Phasianus colchicus*, red-legged partridges *Alectoris rufa* and brown hares *Lepus europaeus* also respond positively to the management regime, and the gamebirds may compete with grey partridges. The GWCT organises some four shoot days a year to reduce their numbers and to offer recompense to participating farmers. It strongly discourages any releasing of reared gamebirds.

The gamekeeper counts grey partridges on both the demonstration and reference areas twice a year, in the spring (pairs, from 2002 onwards) and in the autumn (adults and young, from 2001 onwards). The 2001 autumn counts and 2002 spring pair counts reflect the densities of grey partridges before management began. The 2001 autumn counts on the demonstration and reference areas gave similar low densities, around 8 birds per km^2, and the spring

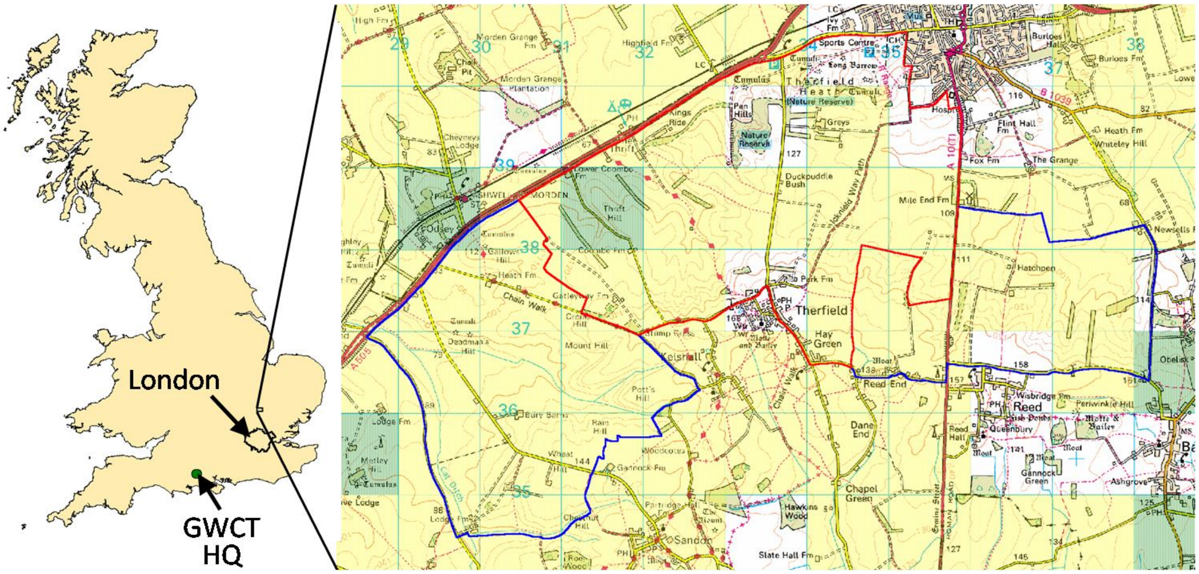


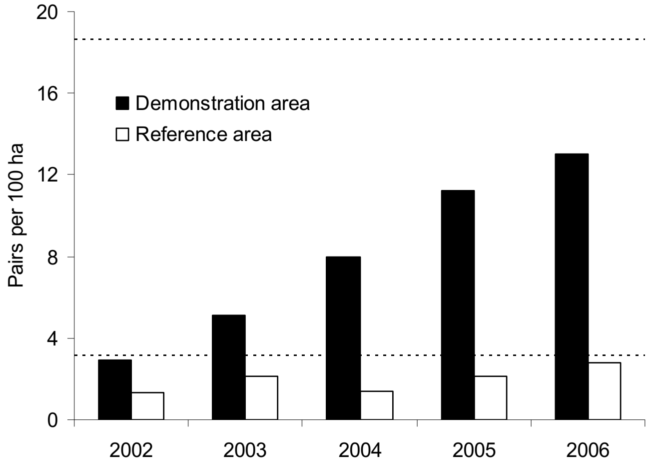
Figure 3: Location of the Grey Partridge Demonstration Project, in eastern England, and Ordnance Survey map outlining the demonstration (blue) and reference (red) areas. Superimposed on the map is a 1x1-km grid, with squares classified as optimal (yellow), suboptimal (green) or unsuitable (white) for grey partridges according to landscape characteristics in 2000 (cf. Figure 2).

densities were below the potential number for unmanaged ground (Figure 4). By spring 2003 the density on the demonstration area had exceeded the unmanaged potential, and by spring 2006 it had increased by 4.5 times, to 13 pairs per km^2. The most recent (2005) autumn count, four years after the onset of management, gave numbers 8 times higher than at the start. On the adjacent reference area, although spring density had doubled by 2006, it remained below its unmanaged potential. The 2005 autumn numbers were less than a third of those on the demonstration area.

Progress Towards the Targets

Based on calibrating the annual BTO index of abundance with the estimated 140,000-150,000 pairs during the 1988-1991 Atlas survey (Gibbons et al. 1993), the present UK population level is around 65,000 pairs (Figure 1). The last four years of data show an ongoing decline, although an increase from 1999 to 2002 means that abundance since 1999 appears stable overall. This contrasts with an average annual rate of decline between 1980 and 2000 of -7%. Optimistically, therefore, the first target in the Species Action Plan may be met. Achieving the second target of 150,000 pairs by 2010, however, looks unrealistic. It requires an increase in the national population of 230% over five years, equivalent to a sustained 18% per annum. Although this has been achieved and more at the local level, as at Royston, the dedication and resources needed to achieve such a result are not typical. If the raising of awareness and the changes in government agricultural policy bear fruit, what is more likely is that the decline will be reversed, with a rate of increase perhaps close to the 8% per annum observed in new Partridge Count Scheme participants (Ewald et al. 2009). Taking 7% per annum to be cautious, compound growth over five years gives an increase of 140%, so the predicted total would be 91,000 pairs in 2010, 127,000 pairs in 2015 and 178,000 pairs in 2020.

It seems clear that Britain will not meet the 2010



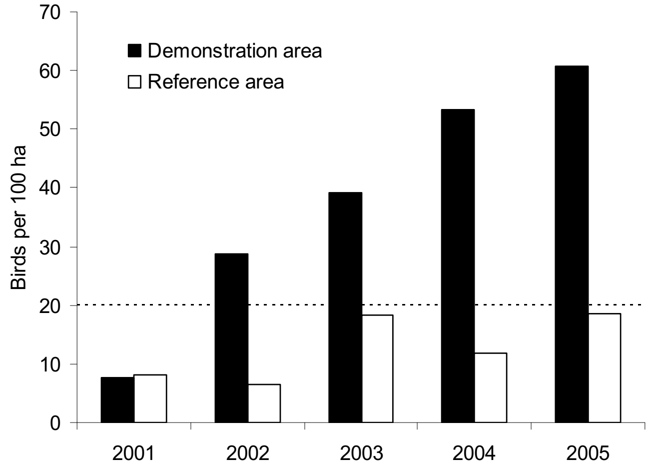


Figure 4: Annual spring (top) and autumn (bottom) grey partridge densities on the demonstration and references areas of the Grey Partridge Restoration Project at Royston. Management began on the demonstration area in January 2002, so the autumn 2001 and spring 2002 counts are representative of an unmanaged situation. The dashed horizontal lines indicate the potential densities based on landscape characteristics for unmanaged and fully managed scenarios.

target set in 1995. Many other species are in the same situation (UK Biodiversity Reporting and Information Group, unpublished). With the aid of its lead partners, the UK government is in the process of revising the original targets in the light of recent knowledge, and extending them beyond 2010. At the same time, biodiversity conservation in the UK is now the responsibility of devolved country administrations, each of which needs its own targets. For the grey partridge, based on the calculations above and taking into account that recovery is likely to slow as numbers increase, the GWCT has proposed that revised UK targets should be 90,000 pairs in 2010, 120,000 pairs in 2015 and 160,000 pairs in 2020 (Ta-

Table 2: Revised targets for the UK grey partridge species action plan proposed to the UK government by the GWCT in 1995 for England, Scotland and Wales, together with practical yardstick values based on the BTO's annual Breeding Bird Survey (BBS) against which to measure success.

	Revised targets			
Country	2005	2010	2015	2020
Abundance (pairs)				
England	55,000	76,000	101,000	135,000
Scotland	9,000	12,500	17,000	22,500
Wales	1,000	1,500	2,000	2,500
Total	65,000	90,000	120,000	160,000
BTO abundance index	0.72	1.00	1.33	1.67
Range (100-km squares)				
England	1,100	1,100	1,150	1,200
Scotland	300	300	325	350
Wales	50	50	55	60
Total	1,450	1,450	1,530	1,610
BBS squares occupied (%)	8	10	12	15

ble 2). It has translated these into country targets by subdividing the totals according to the availability of suitable ground in Table 1.

The government also sought measurable targets for range change. Hitherto, the BTO assessed range change through Atlas surveys every 20 years. More frequent assessments are possible by considering the annual percentage of occupied Breeding Bird Survey squares as a surrogate measure. To determine the current range, we extrapolated from the change in number of occupied 100-km^2 squares observed between the 1968-72 and 1988-91 Atlas periods (England: -14%, Scotland: -24%, Wales: -48%), when abundance fell by 72%, to what the change would be between 1988-1991 and now, when abundance fell by a further 50% (extrapolated change = Atlas change x 50/72). The result, when applied to the range observed in the 1988-91 Atlas, gave a total of 1450 100-km^2 squares occupied by grey partridges in Britain (Table 2). This corresponds to 8% occupancy of BBS squares (average 1999-2003). Since 1994, when the BBS started, the maximum occupancy rate was 14% in 1996 for an estimated population size of 115,000 pairs (from Figure 1). We proposed targets for range expansion that reflected the population targets, taking into account a likely lag between the consolidation of numbers in core areas and recolonisation (it seems probable that numbers would need to build up first and fill gaps within the existing range before noticeable range expansion occurs), and assuming that rates of increase in BBS occupancy would be matched by ones in Atlas squares (Table 2).

Conservation Implications

The decline in numbers of grey partridges in the UK has been so severe that there is no longer any question that the bird must be a top priority for conservation (Gregory et al. 2002). Farmers, land owners and shoot managers are the people on the ground best placed to help restore the fortunes of the grey partridge, and indeed, to benefit if they can increase numbers to a level where sustainable shooting can take place. Education is therefore crucial to raise awareness and encourage such people into sympathetic land management. Until recently, such encouragement was outweighed by harsh economic reality, as the production-driven incentives of the European Union's farm subsidies drove farmers towards ever greater intensification, especially as farm incomes deteriorated after 1995 (e.g., Potts 1997).

Two major recent events, however, have led to a widespread upheaval of UK farming. The Policy Commission on the Future of Farming and Food published a report advising the UK government on how to create a sustainable, competitive and diverse farming sector (Curry 2002), with recommendations for incorporating environmental stewardship into farm policy. A major reform of the European Union's Common Agricultural Policy (Council Regulation (EC) No. 1782/2003) paved the way for breaking the link between subsidies and production. The UK government seized the opportunity to combine both, decoupling production from subsidies with the Single Farm Payment scheme, and tying subsidies instead to good agricultural practice and wildlife-sympathetic land management. It also introduced a new Environmental Stewardship scheme (Anonymous 2005*b*) that replaces and augments previous agri-environmental schemes, and contains many options derived from GWCT research. In England and Wales, these options come in the form of the Entry Level Scheme (Anonymous 2005*a*), open to all farmers who apply, and the Higher Level Scheme (Anonymous 2005*c*), which supports more intensive habitat management with a competitive, targeted approach. The Land Management Contracts offer similar opportunities in Scotland (Anonymous 2005*d*). We have high hopes that these reforms may remove some of the financial barriers that have stood in the way of large-scale adoption of the management required to reverse the grey partridge decline. To showcase how the range of options may best be deployed to aid grey partridge recovery, the GWCT advised farmers from the demonstration area of its Grey Partridge Demonstration Project on choosing and placing Entry Level Scheme options. This Project now offers an essential educational resource that should serve as a source of inspiration for land managers across the country.

In conclusion, the different strands of the GWCT recovery programme form a package that, coupled with the government's agricultural reforms, offers genuine hope for the recovery of the grey partridge in the UK. The tools and targets for monitoring that recovery are also in place, and I look forward to reporting on progress in a few years' time.

Acknowledgments

We thank all the individual farmers, gamekeepers, landowners and scientists, as well as the organisations and companies, who have dedicated time, effort and money to the cause of the grey partridge.

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Conservation Reserve Program Grasslands and Ring-Necked Pheasant Abundance in Minnesota

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Ring-necked pheasant (*Phasianus colchicus*) abundance was measured on 15 study areas using roadside counts during the summers of 1990-1994 to examine possible relationships to permanent grasslands and 9 other cover types. The majority of permanent grasslands was enrolled in the Conservation Reserve Program (CRP) and likely would have been actively used for agriculture if not for the CRP. Roads were divided into 300 m segments and the proportion of each cover type was determined within 200 m and 800 m of each segment. A non-parametric procedure was used to determine the most significant predictors of number of pheasants observed on each road segment during roadside surveys. Year, study area, and proportion of cover type were used as predictor variables. Proportion of permanent grassland cover was the most significant predictor in every model examined. Numbers of pheasants, predominantly broods, were approximately 10 times higher in samples that had $>$30% grassland compared to samples with $\leq$10%. There was no statistically significant increase in number of pheasants as grassland increased from 30 to 100%. Year-to-year variation and differences among study areas were the second most significant factors in predicting the number of pheasants observed. Small grains and pasture were also positively correlated to pheasant numbers. If CRP grassland had not been available, pheasant abundance would have been significantly lower in the study areas.

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Introduction

Populations of ring-necked pheasants (*Phasianus colchicus*) in the Midwest and many other grassland birds have declined in recent decades in response to loss of prime habitat (Warner et al. 1984, Warner and Etter 1986, Sauer et al. 2003). Many investigators have shown that pheasants preferentially nest and raise their broods in non-row crop herbaceous vegetation, especially grasslands, small grains, and hay (Hanson and Labisky 1964, Kuck et al. 1970, Hanson and Progulske 1973, Warner 1979, Ewing 1992). As the amount of row crops increased, both area available for nesting and chick survival rate declined (Warner et al. 1984).

Government farm policies can greatly affect the quantity and quality of habitat available for pheasants. For example, most annual set-aside programs have neutral or negative effects on pheasant populations because fields are frequently left fallow or disturbed during nesting or brood rearing periods (Berner 1984, Kimmel and Berner 1998). However, multi-year cropland retirement programs, with the provision for planting perennial cover, have the potential to reverse pheasant population declines by providing more grassland (Edwards 1984, Berner 1988, Kimmel and Berner 1998). Multi-year cropland retirement programs require participating farmers to remove land enrolled in the program from agricultural production for a set period of time (e.g. 10-15 years for CRP) in exchange for payments. Most

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multi-year cropland retirement programs require the landowner to plant some kind of cover vegetation while the land is in the program. Cropland retirement programs may be established for different purposes including protecting environmentally sensitive land and water, improving wildlife habitat, and reducing commodity production (Napier 1990).

In 1985, the Federal Food Security Act established the Conservation Reserve Program (CRP), which provided for the removal of land from agricultural production and planting of perennial cover for >10 years. Under the CRP, >14 million ha of erosion-prone cropland have been retired in the United States (United States Department of Agriculture 1993). In Minnesota, 96% of CRP land has been planted to cool season (CP1) or warm season (CP2) perennial grassland (Osborn et al. 1992). Common cool season perennial grasses used in the upper Midwestern United States include smooth brome (*Bromus inermis*) and Kentucky bluegrass (*Poa pratensis*) and common warm season perennial grasses include switchgrass (*Panicum virgatum*), Indiangrass (*Sorghastrum nutans*), and big bluestem (*Andropogon gerardii*). Nationally, the majority of these grasslands have not experienced anthropogenic disturbance since being planted, except to control weeds and insect pests (Napier 1990), and for emergency haying or grazing (Hays and Farmer 1990).

It is well established that different vegetation types have different effects on pheasant productivity (Kuck et al. 1970, Gates and Hale 1974, Dumke and Pils 1979, Warner 1979). The relationships of pheasant abundance to differing proportions of vegetation types at varying scales on the landscape has been less well documented, though more recent research has explored different aspects of this issue. Haroldson et al. (2006) demonstrated that for each 10% increase of grass in the landscape, provided primarily by CRP, pheasant surveys averaged 12.4 birds/route higher in spring and 32.9 birds/route higher in summer. Clark et al. (1999) found that pheasants were more likely to nest in areas with more grass and larger blocks of grass. Leif (2005) showed that male pheasants selected areas with grassland and woody cover. The purpose of this study was to explore these relationships at 2 scales: the summer home range of adult pheasants and the home range of 4-8 week old broods. We limited our study areas to those with grasslands primarily originating from the CRP to determine the effect of this program on pheasant populations. We also explore the upper limit where adding more grass to the landscape does not further increase pheasant densities.

Study Area and Methods

In 1990, the Minnesota Department of Natural Resources (MNDNR) began a study to determine the effects of the amount of land in CRP and in a similar state program, Re-Invest in Minnesota (RIM), on pheasant abundance. Fifteen study areas in south-central Minnesota were selected to be as similar as possible except for the amount of CRP/RIM land Kimmel et al. (1992). Topography was flat to rolling and the dominant land use on all study areas was agricultural, with 52-93% in row crops (corn and soybeans). The amount of grassland (primarily CRP) varied from 0 to 30% (Table 1). Study areas were approximately square and between 22-27 km^2. Aerial photographs (1:11000 scale) of the study areas were taken in late July - early August of each year from 1990-1994. Photographs were taken as close to August 1 as conditions would allow. Ground surveys were then performed to identify the land use in each field and note the approximate location of field boundary changes. Using the aerial photos and ground surveys, land use on study areas was digitized using EPPL7, a geographic information system (GIS) Land Management Information Center (1991). Land use in each field was assigned to 1 of 10 cover types: row crops, small grains, hay, pasture (grassland that has been grazed), grassland (undisturbed grassland), wooded areas, buildings, open water, gravel pit, or bare ground. Road systems and watercourses were also digitized. Resolution of the GIS data was 30 m in 1990 and 10 m in 1991-1994. Many features smaller than 30 m across (e.g., roadsides and small bare spots) were not mapped.

Bird abundance was determined using roadside

Table 1: Average percentage of cover types on 15 study areas, south-central Minnesota, 1990-1994. Totals may add to more than 100% due to rounding.

	Study Area														
Land Use	A	B	C	D	E	F	G	H	I	J	K	L	M	N	O
Row crops	91	69	66	82	80	76	75	87	65	74	52	65	65	54	93
Hay	1	2	1	1	3	3	1	1	3	1	1	1	2	5	0
Small grains	1	2	2	2	3	3	2	1	3	3	1	2	2	2	2
Pasture	0	4	1	1	0	4	1	1	1	3	3	1	3	1	0
Grassland	2	15	21	8	6	9	17	6	17	13	30	28	20	29	0
Gravel pit	0	0	0	0	0	0	0	0	0	0	0	0	0	1	0
Woodland	2	3	5	0	1	0	2	1	5	3	6	0	3	3	0
Buildings	2	2	2	2	3	3	2	3	3	3	2	2	2	4	2
Open water	0	1	0	1	2	0	0	0	0	0	2	0	0	0	0
Bare ground	1	2	1	2	1	3	1	0	3	1	3	1	2	0	3

counts Bennett and Hendrickson (1938), Klonglan (1955). Roadside counts have been used to investigate relationships between pheasant populations and habitat Riley (1995). Counts on each study area were repeated 10 times annually between July 20-August 20. Survey routes through each study area were divided into 300 m segments and the number of pheasants within the bounds of each road segment was recorded. We used the number of birds seen on each road segment/10 repetitions as an index of pheasant abundance.

Using the GIS, we calculated the percent of each cover group within 800 m and 200 m in all directions from each road segment. This created roughly elliptical samples with areas of 250 ha (1900 m long by 1600 m wide) and 20 ha (700 m long by 400 m wide), respectively. These sizes were chosen to approximate the summer home range of adult pheasants (250 ha) and the home range of 4-8 week old broods (25 ha) Gates and Hale (1974), Warner (1979), Gatti et al. (1989).

For each road segment sample, total number of pheasants seen was paired with habitat data and analyzed using Formal Inference-Based Recursive Modeling (FIRM), a non-parametric procedure Hawkins (1992). FIRM divides the data into subsets based on the predictor variable that gives the most significant explanation of the variability of the response variable. The output is a dendrogram showing the most significant predictors and how the data were divided according to each of these. Predictors used by FIRM must be <16 classes. For this study we grouped the percentage of each cover type into 11 classes: 0%, >0-10%, >10-20%, etc. Each of the 5 years and 15 study areas was placed in its own class. Thus, each sample was composed of a number of pheasants seen (response variable) and class values for the percentage of each of the 10 cover types, the year it was taken, and the study area on which it occurred (12 predictor variables) Drake (1998).

FIRM calculates the mean and standard error of the response variable for each predictor class. Using a *t*-test, FIRM compares the means of the most similar classes within each predictor. If they are not significantly different, the samples in both classes are merged into a single composite class and the mean and standard error are calculated for the new composite class. This process is continued for all classes within each predictor until no further merging is possible. The result is a grouping of the samples by predictor values. For each predictor, a significance level is computed for the merged groups, based on one-way analysis of variance. Two methods are used to calculate significance of the merged groups. These 2 methods, called the Bonferroni approach and multiple comparison approach, both take into account number of classes in each predictor. This reduces the probability of a predictor being classified as significant simply because it had more classes than another.

The final step in the FIRM analysis is to choose the predictor that has the greatest significance level (smallest *P*-value) and separate the samples into subsets based on that predictor. For example, if Predictor B has 5 classes and was found to be the most significant predictor, it might turn out that classes 1 and 2 were statistically similar to each other and 3, 4, and 5 were similar to each other but different from 1 and 2. The samples in 1 and 2 would be put into a new composite group and those in 3, 4, and 5 would be put into a second group. The analysis is then repeated on each resulting group, using all the predictors again, until no additional significant predictors are found or user specified criteria are satisfied.

Following construction of models using the entire dataset, we performed a model validation step. We randomly assigned each sample to one of three subsets and performed the same FIRM analysis on each subset. The models created for each subset were used to attempt to explain variation in pheasant abundance in the other two subsets. Thus, for each of the four situations (250 ha - all variables, 250 ha - land-cover variables only, 25 ha - all variables, 25 ha - land-cover variables only), three models were constructed and each verified using the other two subsets. This gave 12 models and 24 validations.

Results

We evaluated pheasant abundance on 4275 road segments and their associated 250 and 25 ha samples during the 5-year study. All birds seen were included in the analyses; however, 86.3% were chicks or hens with broods. The average estimated age of chicks seen in this study was 8.5 weeks. The mean number of pheasants seen on each road segment per 10 days of observation was 1.14 (range 0-59). The proportion of grassland in each study area was relatively stable during the 5 years of study and CRP/RIM lands made up 70% of the total amount of grassland.

250 ha Samples

The most significant predictor of pheasant abundance when using 250 ha samples was proportion of grassland (Figure 1). The statistically different groups for 250 ha samples were samples with $\leq$10%, >10-20%, >20-30%, and >30% grassland. Mean number of pheasants seen approximately doubled with each 10% increase in grassland, resulting in 9.7 times more pheasants seen in samples with >30% grassland (group 5) than in those with $\leq$10% grassland (group 2).

Year and study area further influenced pheasant abundance. Year was the most significant predictor for samples with $\leq$30% grassland. Significantly fewer pheasants were seen on samples in groups 2 and 3 in 1992 and 1993, in group 4 in 1993, and in group 3 in 1990. Year was a significant predictor for group 5, also, with a decline in mean number of pheasants seen in 1993. However, study area was more significant in explaining variation within this group. Within group 5, samples in study area B (group 18) and study areas L, M, and N (group 17) averaged 5.8 and 2.4 times as many pheasants, respectively, as seen as in study areas C, F, G, H, I, J, and K (group 16).

When just the 10 cover types were included in the analysis, grassland remained the most significant predictor for the whole dataset (Figure 2). The most important predictor for the lowest grassland group (group 2) was the proportion of small grains. Within group 2, the few samples with >10% small grains averaged 0.98 pheasants per road segment (group 7) while those with $\leq$10% small grains averaged 0.30 (group 6). Within group 6, that is, samples with $\leq$10% small grains, those samples that had >10%

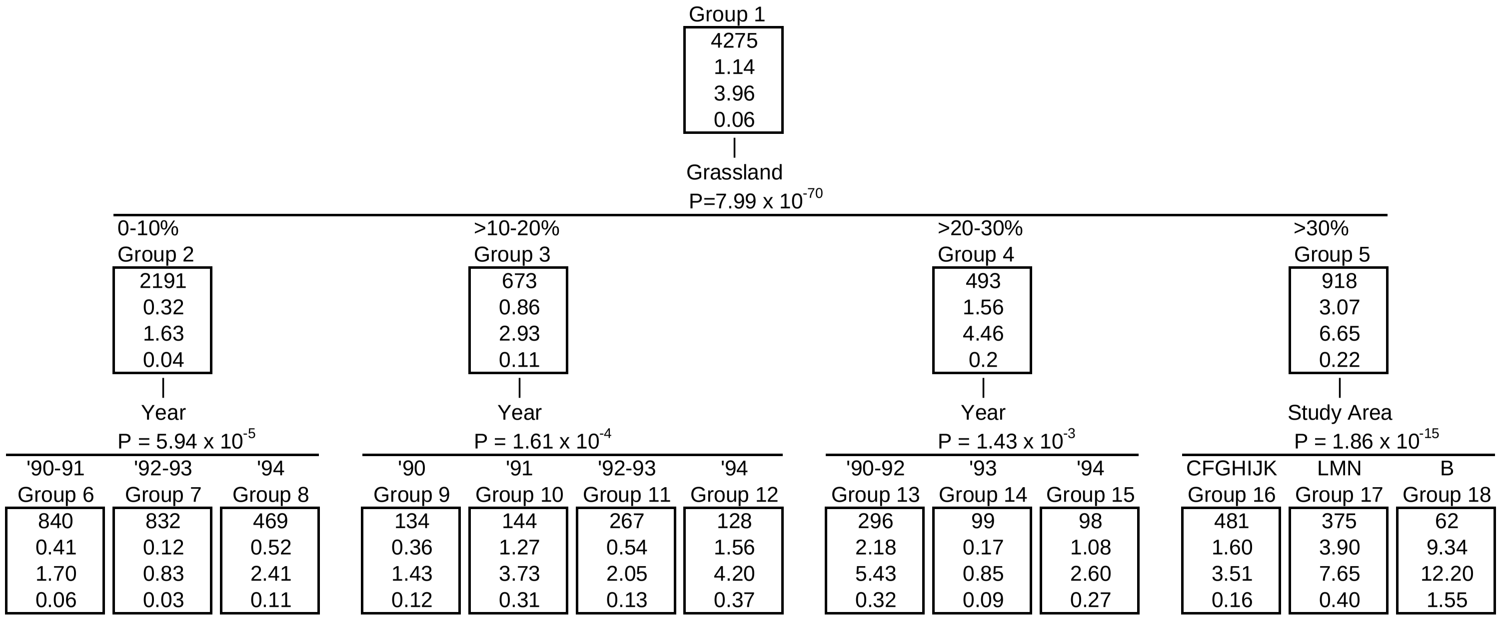

Figure 1: Significant factors affecting ring-necked pheasant abundance on a 250 ha scale, all variables. Most significant predictor of pheasant abundance and P-value are between levels of the dendrogram. Boxes enclose statistics for each significant group. Within boxes, 1^{st} row = number of samples within the group, 2^{nd} row = mean number of pheasants/sample/10 observations, 3^{rd} row = standard deviation, 4^{th} row = standard error.

pasture had 1.1 pheasants/road segment (group 11) while those with ≤10% pasture had 0.29 pheasants (group 10). Groups with >10-20% and >20-30% grassland had no significant predictors for variation in pheasant numbers. Group 5 (>30% grassland) had the most pheasants in samples with no standing water and >10-20% small grains (group 13).

25 ha Samples

The results using 25 ha samples were similar to those for 250 ha samples. The most important predictor of the number of pheasants seen in the 25 ha samples was the proportion of grassland. The significantly different groups were 0%, >0-10%, >10-30%, and >30% grassland (Figure 3). Mean number of birds seen per road segment approximately doubled from group to group. Mean number of pheasants seen in samples with >30% grassland was 9.9 times higher than the mean for samples with 0% grassland. When all predictors were used in the analysis, study area and year were the most significant predictors in all 4 grassland groups (groups 2-5). Study area was a significant predictor of differences in each of these groups. It was the most significant predictor for all samples that had >0% grassland (groups 3-5) and was significant but not the most significant predictor in group 2. Year was most significant in group 2. The effect of study area differences on samples in groups 3-5 was large. Samples within group 3 on study areas with higher pheasant counts (group 11) averaged 5.9 times more pheasants compared to samples on study areas with lower pheasant counts (group 10). Within group 4 the difference was 3.8 times (group 12 vs. 13), and within group 5, the 2 higher study area groups (16 and 17) averaged 4.4 and 9.6 times as many pheasants as the lowest group (14). Year was significant only in the 0% and >30% grassland groups. It was the most significant predictor in the 0% grassland group with significant declines in pheasant abundance in 1992 and 1993. The mean number of pheasants/road segment fell from 0.43 in 1990-1991 to 0.15 in 1992 to 0.02 in 1993 on 25 ha samples with 0% grassland, a decrease of 95%. By contrast, the number of pheasants in the >30% grassland group fell from 3.95 in 1990-1991 to 0.99 in 1993, a 75% decrease. The >0-10% and 10-20% grassland groups experienced non-significant declines in these years. These two groups were smaller than the others and may not have had enough samples to reveal significant year-to-year variation.

When just cover types were included in the analysis, only the >0-10% (group 3) and >30% (group 5) grassland groups were split further (Figure 4). Both were split based on amount of small grains. Group 3 was split into samples that had 0-30% small grains (group 6, 0.6 pheasants/segment) and >30-60% small grains (group 7, 3.8 pheasants/segment). Group 5 was split into two groups: samples with 0% small grains (group 8, 2.6 pheasants/segment) and >0-50% small grains (group 9, 3.9 pheasants/segment).

Model Validation

Each of the models constructed using one-third of the data gave similar results as when the entire dataset was used. The results of model validation for both the 250 ha and 25 ha samples were similar and will be treated together here. In every model based on subsets, grassland was the most significant predictor of pheasant abundance. The points chosen for splitting were different from the model based on all the data, but the trends were similar. The fewest pheasants were seen on the samples with little or no grassland and the number increased with percentage of grassland to 30-40% grassland. Secondary predictors and the groupings based on them varied from model to model, but study area and year were the most common significant predictors after grassland. Hay fields and small grains were the most common secondary predictors when just habitat variables were considered. When the 12 smaller models were validated, the initial splits based on grassland were also significant predictors of the validation data. Secondary predictors were much less consistent in their ability to explain the variation in pheasant abundance.

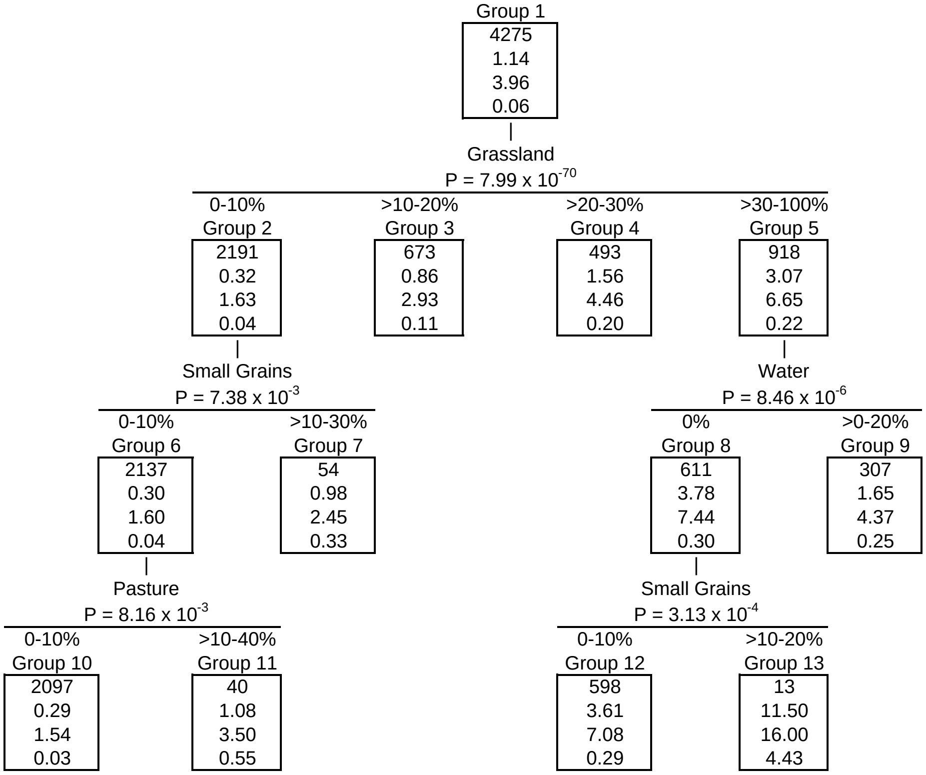

Figure 2: Significant factors affecting ring-necked pheasant abundance on a 250 ha scale, land-cover variables only. Most significant predictor of pheasant abundance and P-value are between levels of the dendrogram. Boxes enclose statistics for each significant group. Within boxes, 1^{st} row = number of samples within the group, 2^{nd} row = mean number of pheasants/sample/10 observations, 3^{rd} row = standard deviation, 4^{th} row = standard error.

Discussion

The proportion of grassland was the most important predictor of the number of pheasants seen. The effect of the proportion of grassland was statistically significant with extremely small P-values. Statistically significant differences may not always be practically significant since, especially with large data sets like we had, small responses within the data can be detected. Our results were also practically significant, though, with a 10-fold increase in pheasant abundance from samples with little or no grassland to those with >30% grassland. No further increase in pheasant abundance was detected in samples with >30% grassland.

The lack of a positive response by pheasants to >30% grassland has two likely explanations. Hanson and Progulske (1973) and Warner (1979) noted that pheasants were found increasingly in row crops after late July. As pheasant chicks shift their diet from exclusively insects to largely vegetable matter after 4 weeks of age, row crop fields and other non-grassland habitats can provide food as the vegetation matures (Loughrey and Stinson 1955). Within the home range of a brood, a moderate amount of grassland cover may provide adequate safe brooding and nesting habitat and beyond that minimum, other cover types may be of equal or greater benefit as greater amounts of grassland. The second possible reason for not detecting any trends in samples with >30% grassland could be that few samples had large amounts of grassland and there was substantial variability ('noise') in the numbers of pheasants

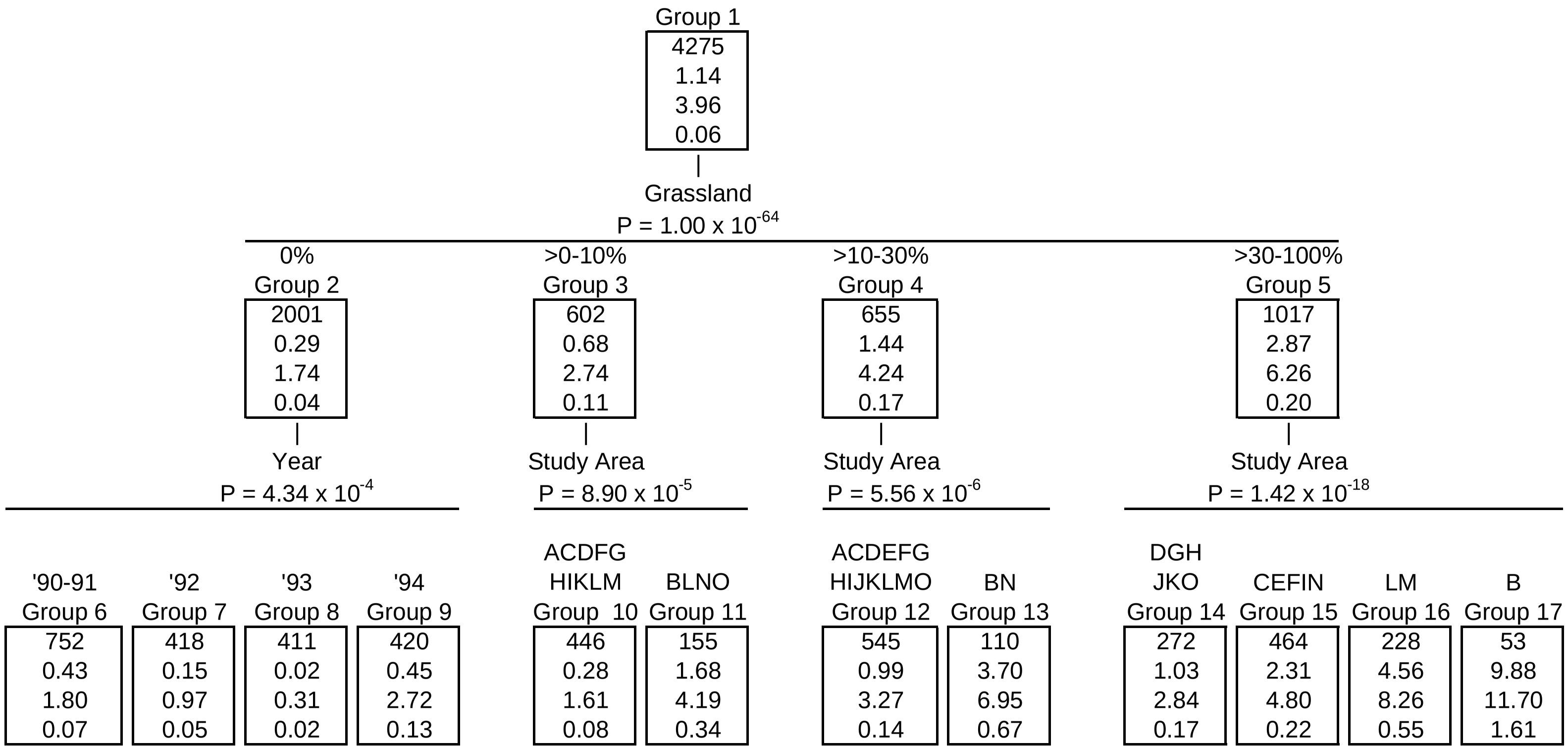

Figure 3: Significant factors affecting ring-necked pheasant abundance on a 25 ha scale, all variables. Most significant predictor of pheasant abundance and P-value are between levels of the dendrogram. Boxes enclose statistics for each significant group. Within boxes, 1st row = number of samples within the group, 2nd row = mean number of pheasants/sample/10 observations, 3rd row = standard deviation, 4th row = standard error.

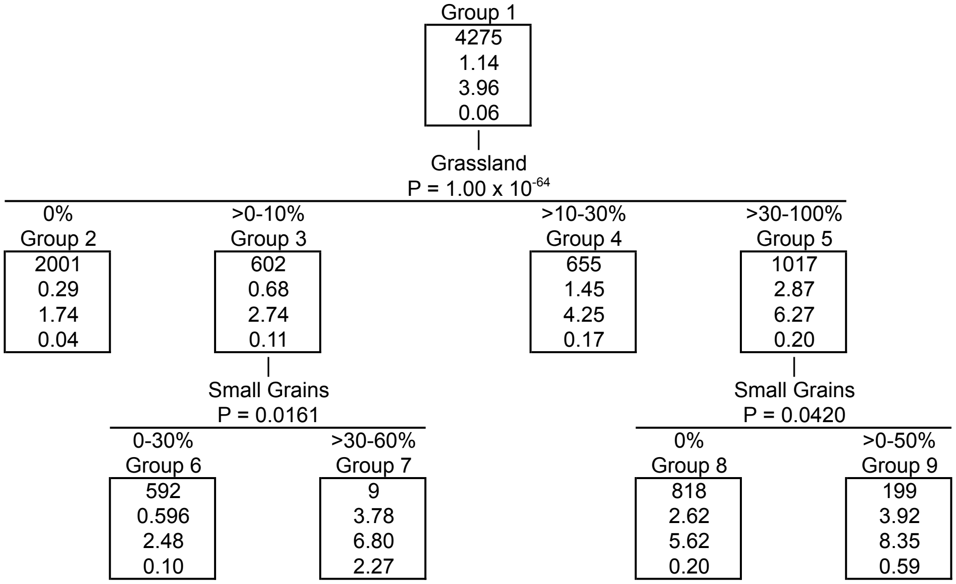

Figure 4: Significant factors affecting ring-necked pheasant abundance on a 25 ha scale, land-cover variables only. Most significant predictor of pheasant abundance and P-value are between levels of the dendrogram. Boxes enclose statistics for each significant group. Within boxes, 1^{st} row = number of samples within the group, 2^{nd} row = mean number of pheasants/sample/10 observations, 3^{rd} row = standard deviation, 4^{th} row = standard error.

seen. For instance, of the 1017 (24% of the total) 250 ha samples that had >30% grassland, only 166 (4% of the total) samples had >60% grassland. Thus, the statistical tests may not have been able to detect differences because the sample size was not sufficient to compensate for the variation.

Habitats besides grassland were also shown to have impacts on the abundance of pheasants. Amount of small grains and pasture were both positively correlated with the number of pheasants seen. These two cover types were not nearly as significant as amount of grassland, but they were important secondary habitats. However, the overall effects of these cover types on pheasant abundance are not clear. These cover types, as well as hayfields, may provide feeding habitat, especially in mid- to late-summer. Warner (1984, pg. 84) stated that hay and small grain fields were "prime pheasant brood habitat". But pheasants attempting to nest in these cover types may be killed by mowing or harvesting, (Warner and Etter 1985). A more complete evaluation of the value of small grains, hayfields, and pasture to pheasant reproduction in southern Minnesota would require study areas with greater acreage of these types. Very few samples had >30% of any of these cover types and the combined total of small grains, hayfields, and pasture was <10% of any study area. These facts limited our ability to detect impacts of different amounts of small grains, hayfields, and pasture.

Other studies have shown the importance of grassland, hayfields, and small grains to pheasant reproduction. Kozicky (1951), Hanson and Progulske (1973), and Warner (1979) showed that small grains and hay were favored vegetation in the summer. Ewing (1992) found that 76% of radio-tracked chicks were located in grass/hay fields although that cover type occupied only 9.5% and 26.5% of his two study areas. Baskett (1947) and Warnock and Joselyn (1964) determined that hayfields, strip cover, pasture, and small grains produced more pheasants than other habitats. All of these studies showed that grassland and grassland-like cover types were more productive and used more for pheasant reproduction than non-grassland-like cover types (e.g. row crops and woody habitat). Of the above studies, only Ewing (1992) included two study areas with differing amounts of grassland, but he did not measure the differences in productivity between them.

Our results imply that pheasant abundance is more susceptible to environmental variation in marginal habitats. The samples with the lowest proportion of grassland were the most affected by yearly variation. In 1990-1991, samples with no grassland had 11% as many pheasants as samples with >30% grassland. In 1993, the figure dropped to 2%. 1993 was an abnormally wet year with an average rainfall on the study areas 46% higher than the other years from May 15 through August 15 (R. Kimmel, MNDNR, unpublished data). Thus, in certain years pheasants can have some success even where there is little grassland. When the environmental conditions are not favorable, the impact may be proportionately greater in landscapes with less grassland. Riley et al. (1998) and Perkins et al. (1997) found similar results in northern Iowa in studies they conducted between 1990-1994. Both studies were done on the same two large (93.2 and 124.3 km^2) study areas. Riley et al. (1998) studied chick survival and found that, although survival rates between the two study areas did not differ in most years, in 1993 the rate was significantly lower in the study area with less perennial grassland. Perkins et al. (1997) examined winter survival of pheasant hens. Although mean survival rates did not differ significantly, variation on the study area with less grassland (9.3%) was greater than on the study area with more grassland (25.0%).

Study area was the most significant predictor in samples that had >30% grassland for 250 ha samples and for all levels of grassland for 25 ha samples. It was significant, but not the most significant, in other groups, also. Certain study areas consistently had more pheasants on samples that had the same proportion of grassland as other study areas. Samples on study areas B, L, M, and N consistently had more pheasants than other study areas. We expected these study areas would have higher numbers of pheasants seen because they had 15-29% grassland and 3 were in the top 5 for the amount of grassland (Table 1). That samples on these areas should differ from samples containing the same amount of grassland and on study areas that overall had equal or greater proportions of grassland is a reflection of factors beyond differences in the proportions of local habitat.

The study area variable is a composite variable. It reflects factors that differ between study areas and occur principally on a larger scale than the samples used in this investigation. These factors include the amount of favorable habitat on a scale larger than the individual samples, mortality/survivorship, and the spatial arrangement of the habitat components in relation to each other. The most important of these factors may be the amount of favorable summer and winter habitat beyond the scale of the individual 250 or 25 ha sample. Favorable summer habitat is necessary for successful nesting and rearing of broods and for survival of adults.

As shown in this study and others (Hanson and Progulske 1973, Warner 1979), grassland is a necessary component in good summer habitat. Large areas with at least moderate amounts of grassland will have more pheasants and any samples in these areas will, on average, also have more pheasants. Samples near productive summer habitats are likely to have more pheasants due to dispersal.

Evidence that the study area variable was not simply a proxy for the amount of grassland on a larger scale was the number of pheasants seen on study area B. This study area had 15.4% grassland, 8th out of 15 in the amount of grassland and only half as much as the highest. Whenever study area was a significant predictor, both 25 ha and 250 ha samples on study area B were in the group with the most pheasants. One possible factor contributing to this difference was the effect of winter habitat. Pheasants prefer different types of cover in summer than in winter and they will move farther between summer and winter habitats than they will within either seasonal habitat (Gatti et al. 1989). Good winter habitat may have been more plentiful near study area B. We were unable to measure the quality and quantity of winter habitat in and around the study areas for this analysis.

A second possible factor is the difference in quality between the types of grass cover or even between individual fields with the same grass cover. We did not separate cool-season grasses (CP1) from warm-season grasses (CP2) in this study. Pheasants have been observed to use CP2 fields more in winter (Delisle and Savidge 1997, R. Kimmel, MNDNR, personal communication), presumably because the CP2 grasses are more resistant to compression by snow.

Another possible contributing factor is the spatial pattern of the cover types. Two study areas with the same percentages of cover types may have different pheasant production because one has an arrangement of habitat types that is more favorable than the other. Although this was not addressed in this study, Gustafson et al. (1994) found that spatial pattern was important for classifying wild turkey (*Meleagris gallopavo*) habitat.

Management Implications

This study demonstrates that 1) proportion of permanent grassland has a great impact on the abundance of pheasants, but this ceases to be statistically detectable at >30% grassland; 2) effects of factors other than summer habitat (i.e. yearly variation and study area effects in this investigation) are also important to pheasant abundance, and 3) cover types other than grassland can be beneficial to pheasants. While managers may not have control over the weather, predation, or other factors that vary from site to site, establishing and maintaining at least moderate amounts of permanent grassland cover is of primary importance in increasing pheasant populations. There will still be substantial year-to-year variation in pheasant numbers but the presence of permanent grassland cover may ameliorate the effects of environmental variation.

Finally, nearly all the grassland examined in this study was the result of CRP/RIM. In the absence of these programs, most samples in this study would have had 0-10% grassland, probably with similar impacts on pheasant abundances (Figs. 1 and 3). Thus, multi-year farm programs, such as CRP, can be an important means of providing permanent grassland cover for pheasants and other wildlife, especially in intensively farmed landscapes.

Acknowledgments

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Methods for Controlling Woody Invasion into CRP Fields in Tennessee

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Woody cover is an important component of northern bobwhite (*Colinus virginianus*) habitat; however, some species such as red maple (*Acer rubrum*) and sweetgum (*Liquidambar styraciflua*) grow aggressively and may become dominant on unmanaged areas. Six treatments with controls were implemented in a completely randomized design on a Conservation Reserve Program (CRP) old-field planted to tall fescue (*Lolium arundinarium*) with extensive invasion by sweetgum, red maple, and other woody saplings to determine the most effective method for reducing coverage of woody plants. Treatments included dormant-season burning in March 2004, applications of triclopyr, imazapyr, and glyphosate in July 2004, mowing in August 2004, and growing-season burning in September 2004. Resulting vegetation structure and composition were measured in July 2005. Percentage woody cover was reduced by all treatments (13-50%) except mowing (65.8%, SE=7.0) compared to control (80.4%, SE=7.6). Imazapyr (13.3%, SE=2.6), growing-season burn (14.2%, SE=3.1), and triclopyr (15.8%, SE=3.5) were most effective at reducing woody cover. Percentage cover of desirable legumes (*Chamaecrista* spp., *Desmodium* spp., *Lespedeza* spp.) was greatest in growing-season burn (54.2%, SE=6.7), imazapyr (28.3%, SE=5.9), and dormant-season burn (24.5%, SE=5.2) treatments. Imazapyr increased coverage of blackberry (*Rubus* spp.), while triclopyr increased coverage of warm- and cool-season grasses. Our results suggest growing-season fire in September was best at reducing woody plants and enhancing habitat for northern bobwhites. Growing-season fire resulted in the greatest coverage of desirable legumes, reduced litter depth, and increased percent bare ground. If burning is not possible, applications of imazapyr or tryclopyr may be suitable alternatives.

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Key words: *Colinus virginianus*, fire, herbicide, northern bobwhite, September burn, sweetgum, woody control

Introduction

Managing woody cover for northern bobwhites (*Colinus virginianus*; hereafter bobwhite) can be difficult. In general, optimal cover for bobwhites consists of an annual weed community for feeding and brood rearing, a perennial grass component for nesting structure, and early successional shrubs to provide cover and mast (Stoddard 1931, Rosene 1969). These plant communities occur along a successional gradient that may be manipulated by management practices (Rosene 1969). While it is widely understood that burning is an effective management practice used to control the structure and composition of early successional vegetation, the effects of season, intensity, and frequency of fire are less clear.

In the southern US, desirable early successional "brushy" cover may include sumacs (*Rhus* spp.), plums (*Prunus* spp.), and blackberries. Unfortunately, "late successional species" (Lorimer 2001) commonly invade fields without proper management. Species such as sweetgum, winged elm (*Ulmus alata*), and red maple do not provide optimal structure for bobwhites and may shade out desirable plant communities. Undesirable woody plants can be controlled by using fire or disking, but these techniques may become less effective once plants advance past the seedling stage. Also, in some areas, use of fire is not a management option.

Advances in forest herbicides may provide managers with a means to control undesirable woody vegetation and improve bobwhite habitat (Miller and Miller 2004, Jones and Chamberlain 2004, Welch

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et al. 2004). Past research evaluated the use of herbicides and fire on wildlife habitat in power line rights-of-way (Arner et al. 1976, Bramble and Byrnes 1976). Several studies examined the effects of forest herbicides and fire used to manage encroaching hardwoods on wildlife habitat in pine stands in the South (Jones and Chamberlain 2004, Welch et al. 2004, Edwards et al. 2004). Research in Mississippi and Georgia tested the effects of forest herbicides for improving bobwhite habitat and controlling bermudagrass (*Cynodon dactylon*) on retired pasture sites (Hamrick et al. 2005, Bond et al. 2005). However, to our knowledge, no studies have examined the effectiveness of herbicide application on reducing undesirable woody plants in CRP fields and compared those treatments with different applications of prescribed fire. The objectives of this study were to determine the effects of late growing-season prescribed fire, dormant-season prescribed fire, mowing, and applications of three herbicides on encroaching hardwoods and resulting habitat for bobwhites in a CRP old-field.

Study Area

We conducted the study on a privately owned 7.3-hectare field in Benton County, Tennessee. The area was sown to tall fescue when it was enrolled into the CRP in 1985. Tall fescue coverage was reduced as undesirable woody species pioneered into the field from an adjacent hardwood stand. The field had been mowed annually since the early 1990s in an attempt to control the invading hardwoods. The portion of the field used in this study was evenly covered by invading hardwoods.

Methods

Study Design

We established treatment plots (0.10 ha) in a completely randomized design with four plots per treatment in March 2004 (28 plots total). Treatments included dormant-season burning in March 2004, applications of 5.60 kg/ha triclopyr (Garlon-4 at 5qts/acre), 0.84 kg/ha imazapyr (Arsenal AC at 24 oz/acre), and 4.48 kg/ha glyphosate (Gly-4 at 4qts/acre) in July 2004, mowing in August 2004, and late growing-season burning in September 2004. Treatment plots were rectangular (15.2 x 67 m) to facilitate herbicide applications. Average flame heights were >1 m and <1 m for March 2004 and September 2004 burns respectively. Herbicides were applied using an agricultural spray coupe with a 15.2-m spray boom using a total solution volume of 200 L/ha. All plots were mowed prior to the study in August 2003 and as a result, vegetation height was conducive to herbicide application in July 2004. Non-ionic surfactant was added to each herbicide application at 0.25% total spray volume to increase herbicide uptake. Control plots did not receive any treatment after mowing in August 2003.

Vegetation Sampling

We measured vegetative response in July of 2005. We measured vegetation characteristics by systematically placing a 1-m^2 subsample plot (Bonham 1989) at 3 locations within each treatment plot. We estimated percent cover of total vegetative canopy, litter, and bare ground, and percent cover of vegetative canopy classes including desirable legumes, other forbs, warm-season grasses, cool-season grasses, brambles, sedges, and woody species to the nearest 5%. Desirable legumes included members of *Desmodium*, *Lespedeza*, and *Chamaecrista*. Legumes considered undesirable (i.e, sericea lespedeza, *Lespedeza cuneata* were counted as forbs. Additionally, we counted the total number of woody stems within each subsample plot and measured litter depth at the plot center. We measured species composition along a 10-m line transect (Canfield 1941) placed along the cardinal azimuth passing through the center of each plot. We measured the distance (cm) along each line transect occupied by each plant species. We identified plants to species when possible. We measured maximum vegetation height at 0, 5, and 10 m along each line transect.

Statistical Procedures

We used a one-way analysis-of-variance (ANOVA) with subsampling error to test for differences in vegetation structure and percent composition among treatments (Montgomery 1997). If F-tests were significant ($\alpha = P < 0.05$), we used Tukey's Honest Significant Difference test to determine if pair-wise differences existed between treatments. All tests were performed using PROC GLM in the SAS system (Littell et al. 2002).

Results

Vegetation Structure

Percent cover was different among treatments for total vegetative cover ($F_{6,21}$ = 5.24, P = 0.002), litter ($F_{6,21}$ = 8.52, $P < 0.001$), bare ground ($F_{6,21}$ = 6.82, $P < 0.004$), forbs ($F_{6,21}$ = 3.14, $P < 0.008$), legumes ($F_{6,21}$ = 7.64, $P < 0.001$), woody species ($F_{6,21}$ = 22.10, $P < 0.001$), cool-season grasses ($F_{6,21}$ = 5.37, $P < 0.001$), warm-season grasses ($F_{6,21}$ = 4.59, P = 0.004), vegetation height ($F_{6,21}$ = 11.77, $P < 0.001$), litter depth ($F_{6,21}$ = 7.62, $P < 0.001$), and total woody stems ($F_{6,21}$ = 7.19, P = 0.003; Table 1). Percent woody cover and number of woody stems were reduced by all treatments except mowing compared to control. Imazapyr, growing-season burn, and triclopyr most effectively reduced percent woody cover. Percentage cover of desirable legumes was highest in growing-season burn, imazapyr, and dormant-season burn treatments. Percentage bare ground was greatest in late growing-season burn. Triclopyr had the greatest cover of cool- and warm-season grasses.

Vegetation Composition

We recorded 47 plant species across all treatments in July 2005. Mean species richness did not differ among treatments ($F_{6,21}$ = 1.28, P = 0.328). We detected treatment effects for sweetgum ($F_{6,21}$ = 20.35, $P < 0.001$), winged elm (*Ulmus alata*) $F_{6,21}$ = 8.73, $P < 0.001$), blackberry ($F_{6,21}$ = 10.43, $P < 0.001$), slenderleaf false foxglove (*Agalinis tenuifolia*, $F_{6,21}$ = 3.76, P = 0.0193), and sericea lespedeza ($F_{6,21}$ = 50.6, P = 0.006). Mean canopy coverage of sweetgum was greatest in control (8.17, SE = 0.55) and mowed (5.26, SE = 0.93) treatments, but was reduced by all other treatments. Imazapyr application increased canopy coverage of blackberry (4.5, SE = 1.01) compared to all other treatments.

Discussion

Woody cover is important to bobwhites (Cram et al. 2002). Taylor and Burger (2000) reported bobwhite broods in Mississippi selectively used habitats with greater canopy coverage of woody species (44.3%) than random sites (21.7%). Bobwhites in Illinois nested in old-fields with 20% woody cover (Roseberry and Klimstra 1984). Cram et al. (2002) observed a threshold-like increase in bobwhite abundance relative to woody cover <2 m; however, Guthery (1999) hypothesized an upper threshold to woody cover likely exists where too little herbaceous cover is present, resulting in a loss of usable space for bobwhites. Welch et al. (2004) defined severe woody invasion in pine uplands in Florida as areas with woody stem densities >5 stems/m^2. Our study area was severely invaded by undesirable woody species and as a result, provided suboptimal bobwhite habitat.

The Northern Bobwhite Conservation Initiative states adequate nesting and brood-rearing habitat often limit bobwhite populations in the South (Dimmick et al. 2002). Suitable bobwhite nesting habitat generally consists of 40-60% vegetative canopy cover of grasses suitable for nesting, 40-60 cm in height (Schroeder 1985). Additionally, bare ground is an important component of bobwhite nesting habitat (Rosene 1969). Triclopyr applications maintained greater warm-season grass coverage than all other treatments. Warm-season grasses, such as broomsedge, provide important nesting cover for bobwhites (Dimmick 1974). Warm-season grasses present on our study site included broomsedge (*Andropogon virginicus*), purpletop (*Tridens flavus*), beaked panicgrass (*Panicum anceps*), low panicgrass (*Dichanthelium* spp.), fall panicgrass (*Panicum dichotomiflorum*), and johnsongrass (*Sorghum halepense*).

Burger et al. (1990) described optimal bobwhite brood cover as fields with diverse annual weed communities produced by recent (<3 years) soil disturbance (i.e., disking or burning). Bobwhite broods

Table 1: Mean, standard error, and Tukeys HSD groupings[a] for vegetation structural characteristics measured in July 2005 for treatments and controls in a CRP old-field in Benton County, Tennessee.

		Control			Bushhog			Dormant-season burn			Late growing-season burn			Imazapyr			Glyphosate			Triclopyr		
	P	*X*	SE		*X*	SE		*X*	SE		*X*	SE		*X*	SE		*X*	SE		*X*	SE	
Cover	<0.001	97.5	(1.7)	a[a]	91.7	(2.1)	ab	90	(2.4)	ab	84.5	(3.4)	bc	81.7	(2)	bc	73.8	(2.7)	c	82.5	(3.6)	bc
Bare	<0.001	0	0	c	1.7	(1.3)	bc	5.8	(1.8)	b	14.5	(3.3)	a	1.7	(1.1)	bc	2.5	(1.3)	bc	1.7	(1.1)	bc
Litter	<0.001	3.3	(1.7)	cd	7.5	(1.9)	bc	4.6	(5.3)	cd	0.8	(0.8)	d	16.7	(2.5)	ab	23.7	(2.7)	a	15.8	(3.4)	ab
Forb	0.006	25.8	(5.6)	b	40	(6.7)	ab	39.2	(7.1)	ab	39.2	(7.1)	ab	62.5	(6.6)	a	55	(6.1)	a	51.2	(8.3)	ab
Legume[b]	<0.001	6.2	(2.9)	c	14.6	(3.8)	bc	24.5	(5.2)	b	54.2	(6.7)	a	28.3	(5.9)	b	15	(4.1)	bc	13.3	(4.1)	bc
Woody	<0.001	80.4	(7.6)	a	65.8	(7)	ab	50.4	(6.1)	bc	14.2	(3.1)	d	13.3	(2.6)	d	32.1	(6.4)	cd	15.8	(3.5)	d
Brambles	<0.001	15	(5.7)	a	15.4	(3.2)	a	19.1	(5.2)	a	19.2	(5.3)	a	28.8	(7.4)	a	13.8	(4)	a	0	0	b
CS grass[c]	<0.001	3.7	(2.5)	bc	7	(3)	bc	4.5	(1.6)	bc	0	0	c	4.5	(1.6)	bc	11.7	(3.1)	b	29.6	(7.2)	a
WS grass[d]	<0.001	5.8	(2.8)	b	7.1	(2.6)	b	4.1	(2)	b	10.8	(3.4)	b	1.3	(1.3)	b	5.4	(3.7)	b	29.6	(8.8)	a
Woody stems	<0.001	4.8	(0.6)	a	4.5	(0.5)	a	2.6	(0.4)	b	1.5	(0.4)	b	1.5	(0.3)	b	2.7	(0.5)	b	1.5	(0.3)	b
Litter (cm)	<0.001	1.5	(0.4)	b	1.8	(0.3)	ab	0.3	(0.2)	ab	0.1	(0.1)	c	1.8	(0.3)	ab	3.3	(0.8)	a	2.3	(0.2)	ab

[a] Means with the same lower case letter(s) within the same row are not statistically different

[b] Desirable legumes

[c] Cool-season grasses

[d] Warm-season grasses

feed heavily on invertebrates (Stoddard 1931); therefore, bobwhite brood habitat quality is directly related to invertebrate availability (Hurst 1972, Jackson et al. 1987, DeVos and Mueller 1993). Although invertebrate availability may be highly variable, greater invertebrate abundance and diversity may be associated with diverse plant communities (Shelton and Edwards 1983), particularly the forb component (Harper et al. 2001). Grass monocultures, regardless of type, support relatively few invertebrates (Fettinger et al. 2002). Availability of invertebrates to chicks is determined largely by vegetation density at ground level, which determines foraging efficiency of chicks (Hurst 1972).

All treatments increased percent forb cover compared to control. No treatment effects were observed in our study for plant species richness, but all treatments met species richness requirements for bobwhite brood-rearing habitat (Schroeder 1985). Increase in desirable legumes by burning and imazapyr treatments likely enhanced brood-rearing habitat (Jones and Chamberlain 2004). Arner et al. (1976) found desirable legume response after burning in power line rights-of-ways was inconsistent and depended on soil fertility and past land use. Bobwhite broods in Mississippi and Florida used areas with mean bare ground cover >20% (Taylor and Burger 2000, DeVos and Mueller 1993). Late growing-season burning produced the greatest percentage cover of bare ground in our study; however no treatment produced enough bare ground to be considered optimal brood-rearing habitat (Schroeder 1985). Although dormant-season burning did not provide the greatest decrease in percentage of undesirable woody cover (the primary objective of this study), the reduced coverage of sweetgum relieved competition on desirable forbs and legumes which likely provided better bobwhite brooding habitat (Jones and Chamberlain 2004).

Growing-season fires have been used to control undesirable hardwoods in pine stands in the South. Rosene (1969) stated growing-season fire would destroy nests, eggs, and broods of birds and should be used only when necessary to control invading hardwoods. Fields with severe woody invasion similar to the one used in our study do not provide suitable nesting or brood-rearing habitat for bobwhites because of a lack of nesting structure, annual plant communities, and adequate bare ground. We recommend growing-season fire in September if bobwhite nesting and brood rearing is limited by undesirable woody encroachment.

While we recommend September burning to manage CRP fields invaded by undesirable woody species; we recognize burning is not always possible. In that case, applications of imazapyr or tryclopyr may provide a suitable management alternative for woody control. Imazapyr may provide greater brood-rearing habitat benefits than triclopyr because it has less adverse effects on legumes and blackberry. Although triclopyr applications resulted in greater coverage of warm-season grasses and controlled sericea lespedeza, cool-season grasses were increased. Bobwhite habitat benefits are reduced when cool-season grasses such as tall fescue (Barnes et al. 1995) and orchardgrass (*Dactylis glomerata*) are present. We recommend managers address invasive plants such as sericea lespedeza, tall fescue, and undesirable woody species aggressively, as the density of these plants will only increase over time. Once invasive plants are controlled, we recommend prescribed fire and disking to set back succession in oldfields and maintain desirable plant communities for bobwhites.

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Management of Montezuma Quail in Texas: Barriers to Establishing a Hunting Season

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Montezuma quail (*Cyrtonyx montezumae*) occur throughout the desert southwest, ranging from Vera Cruz, Mexico to southern Arizona, New Mexico, and western Texas. In Mexico, Montezuma quail are a protected species and in New Mexico and Arizona they are a harvested gamebird. The distribution of Montezuma quail has changed significantly during the past century. Currently Montezuma quail are limited in Texas to the Trans-Pecos with remnant populations in the Edwards Plateau. Although Montezuma quail are classified as a gamebird in Texas, seasons are currently closed. Recently, Texas Parks and Wildlife Department has expressed interest in opening the season on Montezuma Quail. In this manuscript I review and chronicle the sociological and biological barriers associated with opening a hunting season for Montezuma quail in Texas. Sociological barriers include landowner competency and trust in state agencies, a growing voice among non-consumptive users in Texas, a localized ecotourism industry centered on Montezuma quail, and other cultural factors. Biological barriers include lack of data on Montezuma quail population distribution, trends, and abundance; lack of scientific data relative to habitat management for Montezuma quail; or studies simulating the effects of harvest on Montezuma quail. Prior to implementing a hunting season on Montezuma quail in Texas, resource agencies will need to address the biological and sociological challenges outlined in the manuscript.

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Key words: barriers, biological, *Cyrtonyx montezumae*, hunting season, Montezuma quail, sociological, Texas, Trans-Pecos

Introduction

Six species of quail occur in the United States: mountain quail (*Oreortyx pictus*), California quail (*Lophortyx californica*), Gambel's quail (*Callipepla gambelii*), scaled quail (*Callipepla squamata*), Montezuma quail (*Cyrtonyx montezumae*), and northern bobwhite (*Colinus virginianus*). The distribution of the latter 4 species occurs in Texas. Montezuma quail are unique among their counterparts in that they are considered habitat and foraging specialist (Harveson et al. 2007). Montezuma quail are strongly associated with pine-oak woodlands and forage almost exclusively on subterranean foods (*Oxalis, Cyperus*; Stromberg 2000).

All quail species in Texas are classified as gamebirds, but only Montezuma quail have a closed season. In 2003, Texas Parks and Wildlife Department (TPWD) proposed to open the season on Montezuma quail. The proposal was subsequently withdrawn after strong opposition from the public. The hunting public and resource agencies will need to address a myriad of sociological and biological barriers before Montezuma quail are to be hunted in Texas. For this paper, my objectives are to identify and discuss various sociological and biological barriers of establishing a hunting season for Montezuma quail.

Background

History of Montezuma quail in Texas

Montezuma quail occur throughout the desert southwest, ranging from Vera Cruz, Mexico to southern Arizona, New Mexico, and western Texas. The distribution of Montezuma quail in Texas has been greatly reduced where they are currently limited to 6 counties in the Trans-Pecos and 4 counties

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in the Edwards' Plateau (Harveson et al. 2007). Primary causes for the reduction of Montezuma quail distribution and population size is range deterioration. Specifically, land-use practices (livestock grazing) eliminated much of the herbaceous cover and foods that Montezuma quail need to survive in the Trans-Pecos and Edwards' Plateau.

Montezuma quail have had a colorful past in Texas. Early naturalists like Louis Fuertes, John Strecker, and A. Starker Leopold provide some of the earliest literature (Fuertes 1903, Strecker 1930, Leopold and McCabe 1957) on their encounters with Montezuma quail in west Texas and northern Mexico. Following range-wide reduction of Montezuma quail in Texas, several attempts were made to restore their numbers to Big Bend National Park, Guadalupe Mountains National Park, and the Kerr Wildlife Management Area. None of those attempts were considered successful (Harveson et al. 2007).

Hunting in Texas

Although hunting in Texas generates $3.6 billion annually (Brennan 2007), hunting in Texas is in jeopardy (Brown et al. 2003). Quail hunting has also experienced significant changes in the last 2 decades. Adams and Causey (2000) documented a decline in quail hunters from 250,000 in 1988 to 140,000 in 1999. Brennan (2007) noted that quail hunting leases may range from $2/ac in the Trans-Pecos to $12/ac in south Texas. Further, for the first time in Texas, revenue gained from hunting leases has surpassed money generated from cattle leasing (Brennan 2007). Subsequently, quail lessees have opted to have more control over range management conditions of their lease and have purchased grazing leases for their properties.

Proposed open season for Montezuma quail

In January 2003, TPWD proposed to open the season on Montezuma quail with a 2 quail/hunter daily bag limit and a 6 quail/hunter possession limit. Several benefits were anticipated with an open season on Montezuma quail. The first was to minimize incidental take of Montezuma quail by hunters seeking other quails (scaled quail and northern bobwhites). Second, it was thought that TPWD would be able to market a "Texas Quail Grand Slam" (e.g., hunting trips for northern bobwhites, scaled quail, Gambel's quail, and Montezuma quail during the same year) similar to that offered by the agency for ungulates. The Texas Grand Slam is a successful public hunting program and generates considerable income for the agency. Lastly, in times when hunting opportunities appear to be declining, an open season on Montezuma quail could conceivably increase hunting opportunities and possibly increased revenues for landowners.

As with any proposed regulation change, TPWD accepted public comments on the proposed Montezuma quail hunting season via public forums, email, written letters, and phone calls. Five months after the initial proposal for opening a hunting season for Montezuma quail was posted, TPWD withdrew their proposal. The proposal was withdrawn based on public comment (58 against the proposal and 0 for the proposal) and "limited scientific data." Although the initial proposal was withdrawn from further consideration, the resource managers should evaluate possible barriers to changing the season prior to implementing an open season on Montezuma quail.

Barriers to opening a hunting season on Montezuma quail

Sociological barriers

Landowner trust.-Since Texas lands are 97% privately-owned, cooperation between landowners and resource agencies in the conservation and management of game species is essential. Private landowners can ultimately control hunting game species on their property by setting more conservative regulations than those implemented at the state level. In the Trans-Pecos, where Montezuma quail are more prominent, land use trends are changing. Compared to 50 years ago, traditional cattle ranches are less common, ranches are being bought for recreational uses, absentee landowners are becoming the norm, and average ranch size is decreasing (Harveson 2007). Despite the general trends in land ownership and land uses, there is a general distrust of resource agencies in the region. In her essay, Nelson (1991) chronicled the history of distrust which involved issues over the Endangered Species Act, alleged land acquisition, private lands, and a proposed national park. The incidents Nelson (1991) described still affects many landowners today (unpublished data). Further, based on public forums many west Texas landowners are still upset over a season change of mule deer in 1988 and subsequently in 2004.

Non-consumptive users.-As landownership and land-uses in Texas change from agriculture-based to recreation-based (Wilkins et al. 2003), so does its' clientele. Nelle (2002) noted that in west Texas, traditional ranches were being bought for recreational purposes (e.g., hunting and non-consumptive uses). One of the primary non-consumptive uses in west Texas is bird watching. In fact the Big Bend region of Texas boasts a diversity second only to the Rio Grande Valley with >500 species of birds documented for the region (Bryan 2002). Montezuma quail sightings are among the top sightings in the region and generally receive much attention by local birding groups.

Ecotourism industry.-Montezuma quail sighting are so coveted that the Davis Mountain State Park has successfully marketed their local population. Marketing strategies used to lure visitors to the park include a Montezuma quail sightings log at the front desk, photographs of Montezuma quail in various restaurants, caps and bumper stickers of Montezuma quail at their gift shop, and several viewing sites to facilitate Montezuma quail watching and photographing. More recently, other agencies and organizations have realized the successful marketing strategies used by the Davis Mountain State Park. In 2005 Big Bend National Park, which receive 5̃00,000 visitors a year, recently rediscovered Montezuma quail on their property. They subsequently provided press releases to local papers which resulted in a flurry of birders to the Park. In fact, the Park Service is currently evaluating their fire management programs to accommodate Montezuma quail populations. More recently, the Chihuahuan Desert Research Institute of Ft. Davis is planning a viewing site and an interpretative display for Montezuma quail to capitalize on their Montezuma quail sightings. With such wide notoriety, Montezuma quail have been elevated to a flagship species for the Davis Mountains and other sky islands of the Trans-Pecos of Texas.

Time.-One of the biggest challenges the proposed season change faces is time. Hunting regulations are impacted by local cultures and history. With Montezuma quail being protected for >30 years, they have inadvertently become a nongame bird. Although classified as a game bird with a closed season, any changes to their status will be similar to changing a nongame animal to game status.

Biological Barriers

Population dynamics and trends.-Montezuma quail are the least studied quail species in the United States. Although there is a recent interest in the species in Arizona and Texas, no studies have provided empirical data on population dynamics. The lack of data may be attributed to their limited distribution, the inaccessibility of their habitat, and their unique camouflaging ability (Hernandez et al. 2006*a*). Further, Montezuma quail are difficult to monitor. Currently, TPWD does not monitor population trends in Texas as they do with other quails. In New Mexico and Arizona, trends are monitored with a combination of hunter-harvest surveys, line transects, and time-constricted field surveys using trained dogs.

The only trend data available on Montezuma quail in Texas comes from 2 sources: Christmas Bird Count database and the documentation of a range-wide loss of habitat since the 1900s. Although Montezuma quail have been accounted for in Christmas Bird Counts for Texas, their detectability is sporadic leading to no apparent trend. In their review, Harveson et al. (2007) provided historic and current distribution maps for Montezuma quail in Texas. Based on those findings, Montezuma quail populations have decreased substantially since their historic times. They are limited to desert islands in 5 counties in the Trans-Pecos and a core population located in 5 counties in the Edwards' Plateau.

Effects of harvest.-Since little data is available on the population dynamics of Montezuma quail, it is difficult to ascertain at what levels (if any) harvest affects populations. Several researchers have suggested hunting mortality is compensatory (Leopold and McCabe 1957, Brown 1979, Heffelfinger and Olding 2000) but Stromberg (2000) considered Montezuma quail to be vulnerable to overharvesting that may result in possible extirpation of localized populations. Most recently, Bristow and Ockenfels (2000) conducted a study on various aspects of Montezuma quail hunters and harvest. They concluded that: (1) hunters were not concentrating to the level to affect localized populations, (2) hunting pressure and success is low on Montezuma quail with only 2 birds/hunter/day reported in the bag, and (3) the reduction in bag limits from 15 birds/day to 8 birds/day had little effect on the number of hunters or the total number of quail harvest during the season.

Further considerations

If Montezuma quail are, indeed, to be hunted in Texas, additional challenges face the hunting public and TPWD. First, how do landowners market a 2-bird bag limit? Although the distribution of Montezuma quail overlaps that of scaled quail in the Trans-Pecos and northern bobwhites in the Edwards' Plateau their habitats are relatively distinct. The ability of marketing a 2-bird daily bag limit without the added benefit of other gamebirds will be a difficult task and may reap few economic rewards for private landowners.

Second, where will TPWD accommodate public hunting for Montezuma quail? To my knowledge, only 2 state-owned properties in the Trans-Pecos have Montezuma quail: Davis Mountain State Park and Elephant Mountain Wildlife Management Area (EMWMA). Although TPWD provides some public hunting opportunities at Davis Mountain State Park, it is limited to a primitive area for javelina (*Pecari tajacu*) during portions of the archery season. As noted earlier, Davis Mountain State Park has successfully marketed and benefited from viewing of Montezuma quail. Public hunting will likely interfere directly (inability to attract bird to viewing sites) and indirectly (repercussions from nonconsumptive users) with viewing Montezuma quail. EMWMA also has a population of Montezuma quail (Hernandez 2004) and allows public hunting of various game animals (scaled quail, dove, javelina, deer, bighorn sheep). However, EMWMA is the primary brood facility for desert bighorn sheep restoration program (Brewer and Harveson 2006). Hernandez et al. (2006*b*) described Montezuma quail habitat on EMWMA, which coincides with habitat for bighorn sheep (Locke et al. 2005). Thus, it is unlikely that DMSP and EMWMA personnel will be willing to open public hunting for Montezuma quail.

Ultimately, the question to be asked is "Does Texas have harvestable populations of Montezuma quail?" One can not surmise from current knowledge of Texas populations, that Montezuma quail are a renewable resource that can sustain harvest. Lack of data on population status, trends, and basic life history information on Montezuma quail prohibits TPWD from changing their current status. In fact, Cooke (2007) noted that "the hunting seasons on all game animals and game birds in Texas are closed by the Legislature unless data collected by TPWD suggests that a hunt can safely be conducted on a species and this suggestion becomes a finding of fact by TPW Commission."

If Montezuma quail are to be hunted in Texas, research on their ecology will have to be prioritized. Harveson et al. (2007) addressed research needs for Montezuma quail in Texas as: (1) documenting basic biological data (survival, density, habitat use, and movements); (2) establishing and evaluating a monitoring protocol for population trends; (3) using conservation genetics to determine movements and effects of habitat fragmentation on subpopulations; (4) establishing a reintroduction protocol for efforts to restock in formerly extirpated habitats; (5) using population modeling to evaluate the effects of harvest on population dynamics; and (6) exploring the human-dimension aspects of opening a season on Montezuma quail (potential consumptive and non-consumptive users).

Conclusion

If Montezuma quail are to be hunted in Texas, the hunting public and TPWD will need to address a variety of sociological and biological issues prior to implementing a hunting season on Montezuma quail. Foremost, biological data is needed to better understand the population dynamics of Montezuma quail and the potential impacts of harvest on population sustainability. Second, since Montezuma quail have not been harvested in Texas for >30 years and they are the focal species for a strong ecotourism industry, it would be difficult to for the general public to accept an open season for Montezuma quail. These sociological perceptions have inadvertently elevated Montezuma quail to "nongame" status.

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Long-Term Climate Trends and Northern Bobwhite Populations in South Texas

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Because climate change and its associated weather changes may influence population trends of birds, we analyzed northern bobwhite (*Colinus virginianus*; bobwhite hereafter) age ratios and abundance in relation to climate trends during 1908-1997 in south Texas. Weather variables included regional mean temperature maxima for June, July, and August, and precipitation totals for autumn (Sep-Nov), winter (Dec-Feb), spring (Mar-May), and summer (Jun-Aug). Long-term temporal trends for these weather variables were estimated with a linear regression. Yearly weather data were used to predict bobwhite age ratios (juv/ad in autumn and winter) and abundance between 1908 and 1997 using neural network models. We compared these predictions with data available from various bobwhite surveys in south Texas over the period 1940-1999. Means for daily maximum temperature during summer declined at rates between 1.6 and 2.3°C/century. No temporal trends were detected for seasonal precipitation (1908-1997), age ratios (1940-1999), or abundance (1977-1998). Neural models developed independently to predict bobwhite age ratios and bobwhite abundance from weather data produced predictions that were consistent with each other. Years with high age ratios tended to coincide with or precede years of high abundance.

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Key words: climate trends, *Colinus virginianus*, northern bobwhite, south Texas

Introduction

Researchers have demonstrated that global climate change may be associated with long-term trends in bird populations (Root 1993). There also is evidence that the population dynamics of some avian species, such as northern spotted owls (*Strix occidentalis caurina*), might be governed by weather patterns independent of habitat (Franklin et al. 2000). In England, 31% of bird species (n = 65) exhibited significant trends towards earlier nest-initiation over a 25-yr period (Crick et al. 1997). These changes in nest-initiation date were related to climate change for 38% of the species exhibiting long-term trends (Crick and Sparks 1999). In the United States, the mean Julian date for the initiation of the first clutch of the Mexican jay (*Aphelocoma ultramarina*) advanced by 10.1 days between 1971 and 1997 (Brown et al. 1999). Klimstra and Roseberry (1975) reported that the length of the bobwhite nesting season declined by 12 days for each 1°C increase in mean July-August maximum daily temperature. Climate change is thought to be a leading cause of the decline of capercaille (*Tetra urogallus*) in Scotland between 1975 and 1999 (Moss et al. 2001). Similarly, Guthery et al. (2000) hypothesized that global climate change could reduce the percentage of female bobwhites that attempt to lay and the number of nesting attempts per hen, which could lead to reductions in annual production.

The northern bobwhite's range extends through most of the eastern and central parts of the United States (Kaufman 1996). Because bobwhites are an important game species across their range, bobwhite demography is relatively well studied (e.g., Roseberry and Klimstra 1984) compared to other avian

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species. Bobwhite populations tend to exhibit boom-or-bust dynamics (Stoddard 1931, Roseberry and Klimstra 1984, Peterson 2001), and weather patterns may be a determining factor in such dynamics. Because they are so well studied, bobwhites offer a unique opportunity to evaluate the impacts of climate change on their life-history characteristics.

Our aim was to investigate patterns in bobwhite demography with respect to weather patterns. To that end, we examined long-term trends in summer (Jun, Jul, and Aug) temperature maxima and seasonal (fall, winter, spring, and summer) rainfall in south Texas and compared these trends with bobwhite age-ratios and abundance. If climate patterns strongly influence bobwhite population dynamics, then we expected age ratios and bobwhite abundance to correlate with climate patterns within the same timeframe. Finally, we used neural network models to predict bobwhite age ratios and abundance between 1908 and 1997 based on weather records from south Texas. Previous research on bobwhites demonstrated the importance of weather conditions on abundance (Lusk et al. 2002) and production (Guthery et al. 1988, 2002). We used the predictions of the age ratio model to predict when boom years should have occurred within the historical record and corroborated these predictions using the results of the abundance model.

Study Area

Data used in these analyses were obtained from the South Texas Plains. This ecoregion has a level to rolling topography with elevation ranging from ∼305 m to sea level (Gould 1975). The South Texas Plains were originally a grassland savannah, but now contain a significant shrub component due to heavy grazing. This process is well established in desert grasslands (Bahre 1995). Age ratio data came from the Chaparral Wildlife Management Area (CWMA), a semiarid thornscrubland (Gabor et al. 2001) located in Dimmit and LaSalle counties of south Texas, and encompassing 61.5 km^2. Rainfall in the South Texas Plains varies between 40.6 and 88.9 cm annually (Gould 1975), while at the CWMA rainfall averages 64 cm annually (Hellgren et al. 1995, 2000).

Methods

We obtained data (EarthInfo, Inc. Boulder, Colorado, USA) from the weather station located in Falfurrias, Texas, which is roughly centered in the South Texas Plains. Records were available for the period 1908-1997, but were not collected continuously over this period. Whenever possible, we substituted data from the next nearest weather station (Carrizo Springs) when weather data for a particular year were unavailable from the Falfurrias station. Carrizo Springs is 224 km northwest of Falfurrias in the northwestern South Texas Plains. However, in some years, weather data were unavailable for both stations. Therefore, we had data for 79 of a possible 89 years. These data consisted of South Texas mean temperature maxima for June, July, and August, and precipitation totals for fall (Sep-Nov), winter (Dec-Feb), spring (Mar-May), and summer (Jun-Aug). We smoothed the time series for each variable with a 3-point moving average (Kendall and Ord 1990). Estimates of trend in weather variables through time were determined from the slope of the simple linear regression equation for time (year) regressed on each weather variable separately.

We also used linear regression to estimate trends in age ratio and abundance data. Age ratios were obtained from Lehmann (1984) and from records of the CWMA. These records encompassed the period 1940-1999 (n = 46 records for 60 years), but, as with the weather data, some years were missing from these data. Abundance data were obtained from Texas Parks and Wildlife Department's annual roadside quail survey (Peterson and Perez 2000). These records cover the period 1977-1998 and are continuous over the period.

Guthery et al. (2002) used a portion of the above described weather data to develop a neural model to predict bobwhite age ratios. Neural modeling is a powerful analysis tool for describing functional relationships between a set of predictors and responses. Further information on this modeling technique can

Table 1: Slope (β = point estimate) and 95% lower (LCL) and upper (UCL) confidence limits for the linear relation between weather variables and year in south Texas, 1908-1997.

	Slope		
Period	β	LCL	UCL
Mean maximum temperature (°C)			
June	-0.023	-0.037	-0.009
July	-0.016	-0.028	-0.004
August	-0.022	-0.034	-0.009
Seasonal precipitation (mm)[a]			
Fall	0.406	-0.739	1.551
Winter	0.386	-0.182	0.954
Spring	-0.015	-0.715	0.685
Summer	0.062	-0.828	0.953

[a] Fall = Sep-Nov, winter = Dec-Feb, spring = Mar-May, and summer = Jun-Aug

be found in Smith (1996) and a discussion of ecological applications of this method can be found in (Fielding 1999). Specifically, the neural model was developed using age ratio data from Lehmann (1984, :133; 1940-1972) and CWMA records (1973-1997) and the appropriate years of weather data (1940-1997), and contained 7 inputs (independent variables), 2 hidden nodes, and 1 output (age ratio). The model (Guthery et al. 2002) was developed using 80% of the data (n = 28) and the remaining 20% were used for model testing (n = 7). We used this model to predict age ratios over the period 1908-1999, and the accuracy of these predictions was determined by comparing predicted age ratios with population abundance estimated from roadside counts (Peterson and Perez 2000). If the models performed acceptably, we would expect a crude correlation between age ratios and abundance because juveniles normally make up 50% of the autumn population. Such comparison allowed us to examine patterns of population booms based on production.

Bobwhite abundance was estimated from the above weather data in an analogous manner using a neural model developed for Texas Parks and Wildlife Department to predict fall abundance from weather and land-use data (Lusk et al. 2002). We used the predictions of this second neural model to examine population booms based on production as estimated from the age ratio model and to assess the overall performance of the two models. We did this by comparing the agreement with regard to increases and decreases in bobwhite productivity between the two models. Because the dependent variables used in the two models were different (abundance vs. age ratios), and were gathered at different times and in different locations within the South Texas Plains, this comparison represents a true validation of the model (Conroy 1993, Oreskes et al. 1994, Conroy et al. 1995). However, the neural model developed by Lusk et al. (2002) had 3 variables in addition to those used in the model of age ratios (Guthery et al. 2002): 1) proportion of county area in cultivation, 2) head of cattle per ha of non-cultivated land, and 3) previous year's bob-

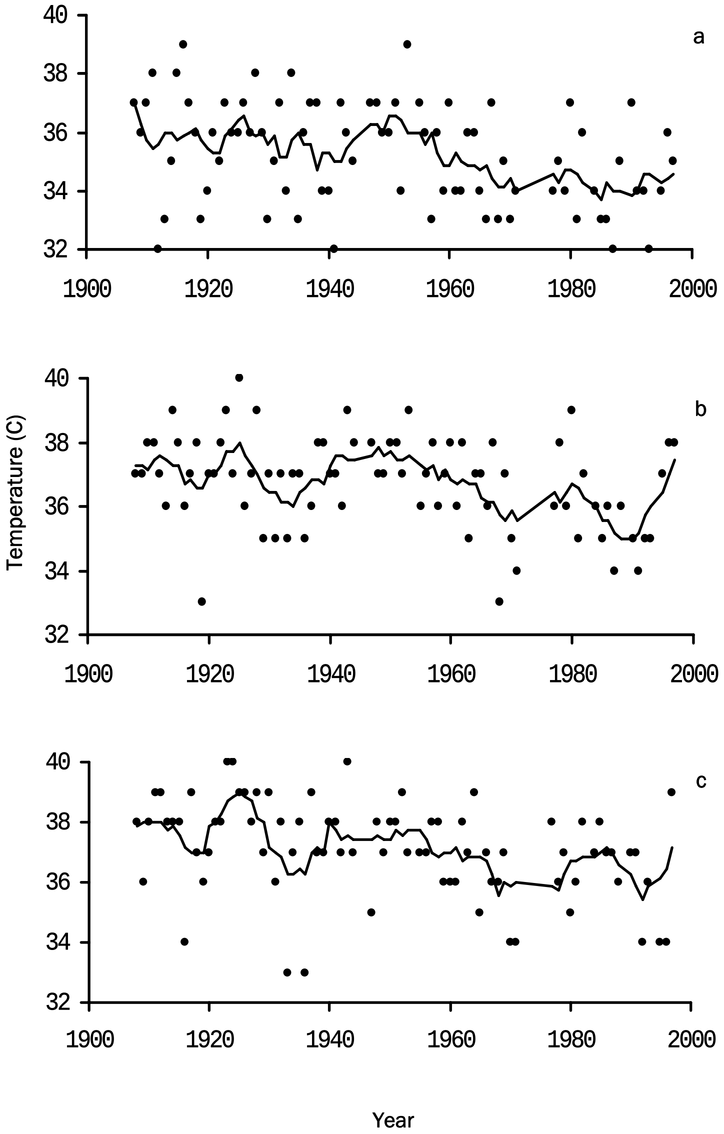

Figure 1: Mean of daily temperature maxima and smoothed trends (3-point moving average) for (a) June, (b) July, and (c) August in south Texas (Falfurrias station), 1908-1997.

white count. We used long-term weather data from south Texas to predict annual bobwhite abundance from 1908 until 1997 using this model (Lusk et al. 2002). Since we were interested only in the effects of weather, we held the values of the other three variables constant at their long-term means, and, therefore, the results we obtained are due to variation in weather only. By holding these variables constant at their mean, we assumed that this would not influence the overall dynamics of the bobwhite population response to weather variables. We tested this assumption by varying the value used for these variables. In all cases, the subsequent dynamics were similar and only changes in magnitude were observed. Therefore, we report results only for the analysis in which cultivated land, livestock density, and previous year's counts were held at their long-term means.

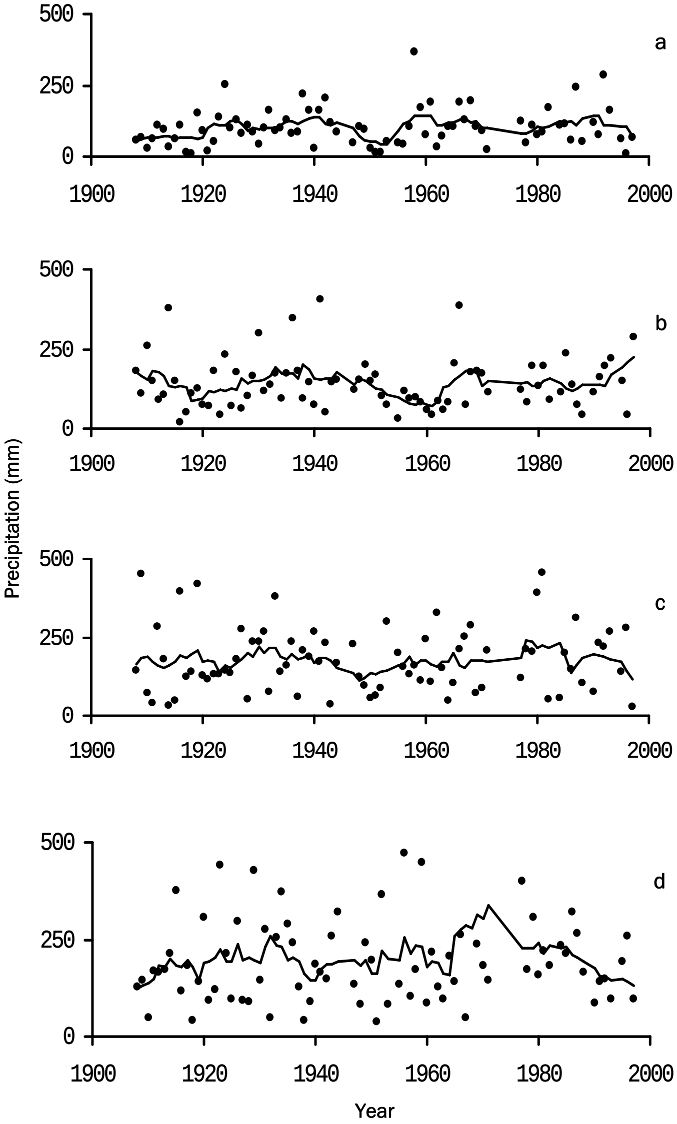

Figure 2: Seasonal precipitation and smoothed trends (3-point moving average) for (a) winter, (b) spring, (c) summer, and (d) fall in south Texas (Falfurrias station), 1908-1997.

Results

Long-term weather patterns

The means of maximum daily temperatures for the summer months (June, July, and August) exhibited a long-term, but highly variable, decline between 1908 and 1997 ($P < 0.05$; Table 1, Figure 1). The rate of decline ranged between 1.6 and 2.3°C/century (Table 1). However, there was no apparent long-term trend in seasonal precipitation ($P > 0.10$; Table 1, Figure 2). Regression coefficients for all seasons indicated a change in magnitude of precipitation of only a fraction of a millimeter per century (Table 1). Further, the 95% CLs for precipitation in all seasons overlapped zero.

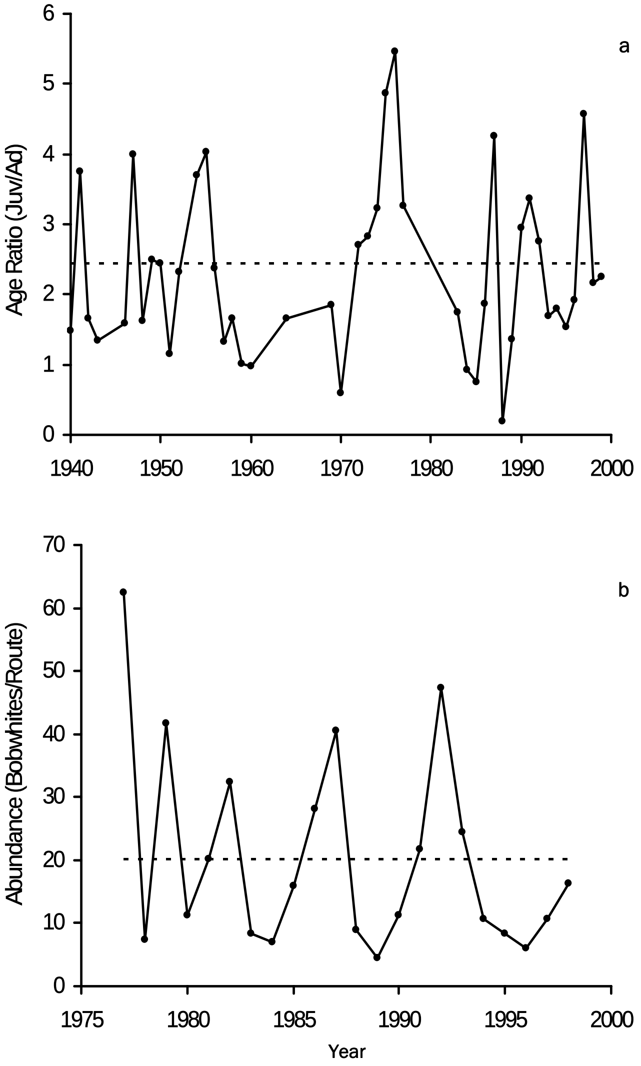

Figure 3: Empirically observed (a) age ratios and (b) abundance of northern bobwhites in south Texas. Dashed line in (a) indicates the mean age ratio of 2.45 juv/ad (1940-1999), and in (b) indicates the mean abundance of 20.22 bobwhites/route (1977-1998).

Bobwhite population trends

During 1940-1999, bobwhite age ratios at CWMA exhibited considerable variability through time (CV = 52.7%; Figure 3a). Further, the linear functional relationship was y = 2.1 + 0.003x (P = 0.753), indicated no temporal trend in age ratios. The intercept was close to the mean for the dataset (2.45 juv/ad) and the 95% CLs encompassed zero. Similarly, bobwhite abundance in the South Texas Plains between 1977 and 1998 exhibited considerable variability through time (CV = 71.0%; Figure 3b) and no temporal trends evident in the time series (y = 29.5 - 0.809x; P = 0.129). The intercept was higher than the mean abundance (20.2 bobwhites/route) for the South Texas Plains over the same period.

Neural model projections

Assessments of model performance are provided in Guthery et al. (2002) and Lusk et al. (2002). Neural model predictions of age ratios based on weather

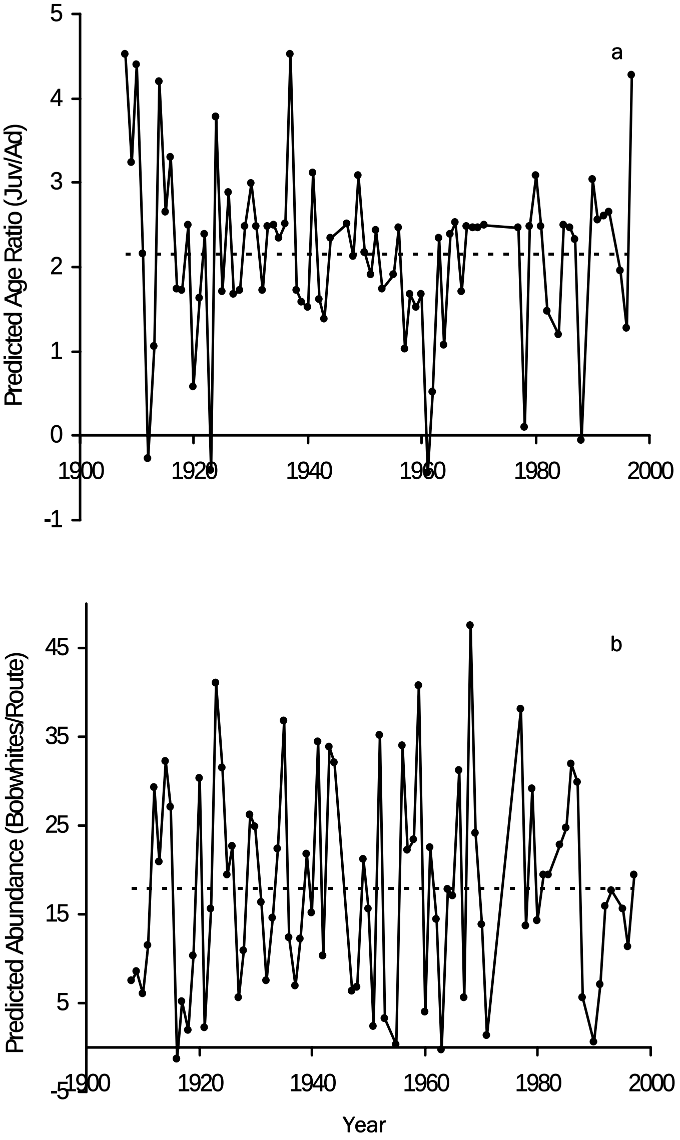

Figure 4: Neural model predictions of northern bobwhite (a) age ratios and (b) abundance in south Texas based on weather variables. The horizontal line in (a) indicates the mean age ratio of 2.15 juv/ad and in (b) indicates the mean predicted abundance of 17.88 bobwhites/route.

variables showed no apparent trend through time ($P = 0.41$) but were characterized by high variability (CV = 47.7%; Figure 4a). An analysis based on abundances from roadside counts (Peterson and Perez 2000) indicated that population booms were associated with 3-4 years of above-average bobwhite production. Therefore, we expected population booms to occur with $\geq$3 years of above-average age ratios, as predicted by the neural model. Based on this criterion and the neural model projections, boom years were expected in 1910, 1916, 1931, 1970, 1981, 1992, and 1993. However, abundances predicted from the neural model indicated boom years (2 mean abundance, or $\sim$40 bobwhites/route) in 1923, 1959, and 1968 (Figure 4b). Further, except for 2 cases, all of the years expected to have population booms based on age ratios were below mean predicted abundance (20.2 bobwhites/route; Figure 4b). That is, the years for which booms occurred in south Texas generally had lower than average predicted abundance. Although our expectations of the occurrences of boom years were not borne out by the abundance predictions, our subjective observation of the 3-4 years of predicted above-average production preceding a boom year may have been spurious. Further, years of high abundance tended to co-occur or follow years of above-average production, and trends tended to change in the same direction (i.e., when age ratios increased from one year to the next, abundance also tended to increase from the previous year) (Figure 4b).

Discussion

Our analysis of bobwhite demography relative to weather variables in south Texas was subject to sources of uncertainty and bias that require explanation. First, the weather variables used in modeling exert variable influence on quail demographics in semiarid environments. Precipitation occurring in association with reproduction has a more powerful influence on productivity than precipitation occurring in seasons that precede reproduction (Heffelfinger et al. 1999, Guthery et al. 2002). Thus, spring precipitation was the most meaningful and fall precipitation the least meaningful precipitation variable in our study; Kiel (1976) also found a strong correlation between spring precipitation and bobwhite productivity in south Texas. Guthery et al. (2002) observed a positive influence of June temperature maxima on the age ratio. They also observed that the age ratio was insensitive to July temperature maxima up to a threshold value of about 36°C; production collapsed above this temperature. August temperature maxima were relatively inconsequential because the reproductive season either had ended or was in sharp decline during August in south Texas (Guthery et al. 1988). We note, also, that temperatures and seasonal precipitation apparently interact in a complex, nonlinear manner in modulating the production of Gambel's quail (*Callipepla gambelii*) (Heffelfinger et al. 1999) and bobwhites (Guthery et al. 2002) in semiarid environments.

Second, the neural model predictions of age ratios and population trends were subject to several problems. The use of data from the Falfurrias or Carrizo Springs stations operated under the strong assumption that data from these stations were representative of the conditions throughout south Texas. Moreover, the weather variables used, though empirically supported in some cases, were arbitrary. Our modeling on means (temperature maxima) and seasonal totals (precipitation) did not account for any catastrophic events such as deluges or severe heat waves, or for the lag effects of rainfall on vegetation.

Third, the dependent variables (age ratio, counts) were themselves subject to uncertainty and bias. The age ratio is a function of reproduction phenology, the proportion of hens that lays, nest success, survival rates for chicks and adults, propensity to renest and multiple-brood, and length of the laying season (Guthery and Kuvlesky 1998). Weather undoubtedly influences these demographic variables in a complex manner. Counts of bobwhites from roadsides, the index used to judge population trends, also are suspected of bias (Lusk et al. 2002). For example, drought may force bobwhites to roadsides that have cover, thus biasing counts upward in drought years. Conversely, periods of precipitation may permit the general use of habitat space away from roadsides, thus biasing counts downward in rainy years. The comparative quantity of vegetation in rainy and drought years also would influence the visibility of coveys and individuals (Lusk et al. 2002).

Despite the problems mentioned above, the neural models revealed apparent patterns in the data (see Guthery et al. 2002). Model predictions reflected the high variability in production and abundance that characterizes quail populations in semiarid environments. The models also provided an approximation of historical trends in production and abundance. As such, we can conclude that the model captures the dynamics of bobwhite populations in this region of their range.

In conclusion, our analysis contained weather variables (spring precipitation, June and July temperature maxima) that have empirical support as correlates of bobwhite production in semiarid environments. Whereas there was evidence of declining temperature maxima during 1908-1998, the rate of decline and the total decline over the period of record were small and probably not biologically significant for bobwhites. There was no trend in spring precipitation. Therefore, in accordance with the models in Kiel (1976), there was no trend in production or abundance of bobwhites, whether these trends were based on empirical data (1940-1997 for age ratios, 1977-1998 for populations) or on neural model predictions before and during the period of empirical record. Global climate change did not appear to be influencing bobwhites in south Texas during 1908-1998. However, as climate change progresses, this assessment will have to be reevaluated.

Acknowledgments

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Hunting Success on Albany, Georgia Plantations: The Albany Quail Project's Modern Quail Management Strategy

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The Albany Quail Project began in March 1992 as a cooperative between Auburn University's School of Forestry and Wildlife Sciences and the private quail plantation community surrounding Albany, Georgia. The goal of this collaboration was to increase population densities and hunting success of northern bobwhites (*Colinus virginianus*) on these properties. Issues important to the plantations were investigated via field research techniques (e.g. radio-telemetry) and then modifications in management regimes were put into practice. From 1992-2006 over 8,000 wild bobwhites were radio-tagged to study various demographic and population mechanism of bobwhites. The result has been a change in management style based on science creating the concept of Modern Quail Management (MQM) techniques. Bobwhite population densities and hunting success during the last decade on these properties have been higher and more stable than witnessed during any previous period in their history. On our primary study site during 1980-1996 the average number of coveys seen per hour was 3.9 (SE = 0.265) with a range of 2.7 - 6.5 (CV = 0.28). Following the implementation of MQM techniques this average during 1997-2005 increased to 7.7 (SE = 0.317) with a range of 6.9-9.7 (CV = 0.12) and population density has been maintained at or near 5 birds per ha. This manuscript provides a brief history of the Albany Quail Project, describes how results from these studies have contributed to the evolution of new management philosophies over the last decade, outlines what these changes have been, and describes how they have been successfully applied on numerous properties in the Albany area to improve bobwhite populations and hunting success.

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Key words: *Colinus virginianus*, Georgia, hunting, modern quail management, northern bobwhite, plantation, radio-telemetry

Introduction

The decline of northern bobwhite (*Colinus virginianus*) populations across most of the southeastern United States has been well documented (Sauer et al. 2004). The decreasing number of bobwhites caused concern among quail biologists and quail enthusiasts for many years with various reasons for the decline discussed and remedies debated (Brennan 1991). No firm answers to the problem existed because the reasons for quail declines were numerous and interconnected. Socio-economic change had occurred across the southeastern landscape since the initial research related to quail management conducted by Stoddard (1931) and Rosene (1969). Modern agricultural and forestry practices have become much more efficient during the latter half of the 20th century and are generally detrimental to quail populations.

Despite considerable changes in the landscape and the factors affecting quail populations, approaches to quail management remained relatively unchanged as little new research into quail ecology was being conducted. No comprehensive, long-term studies were underway to assess quail management techniques in relation to changes in the southeastern landscape and the associated declines in quail numbers during the 1970s and 1980s.

Quail hunting has a strong tradition in South Georgia, and since the early 20th century large land holdings have been managed to maximize quail pro-

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duction for this purpose. Because of this objective, intensive modern agricultural and forest management was not as readily adopted on these properties compared to the rest of the Southeast. Consequently, populations on these privately owned lands remained relatively stable compared to the rest of the southeastern U.S. (Brennan et al. 2000, Palmer et al. 2002). However, from the late 1970s through the early 1990s even these South Georgia quail plantations, the last stronghold of wild quail in the state, were beginning to experience declines and concern mounted.

Along with the strong quail hunting traditions of this area there has been a long history of landowners supporting quail research and allowing properties to be used as outdoor laboratories for research on quail ecology and management. The work of Stoddard (1931), Rosene (1969), Simpson (1976), and Tall Timbers Research Station (Landers and Mueller 1986) are some examples of research efforts made possible by the area's private landowners in the past. When covey finds began to significantly decrease even in the face of intensive management activities, these owners and managers decided that new research based techniques were needed to determine weak points in traditional habitat management. One of the first current programs to address the quail decline with a comprehensive approach of research, land management, and outreach began in 1992 by Auburn University on an intensely managed quail property near Albany, Georgia. While quail hunting success among these sites was still relatively good, records showed it was not equal to some of their better years in earlier decades. The feelings of the landowner and manager were that the traditional management program utilized was ineffective, but it was unclear why. The landowner agreed to fund a 3-year research project to determine what was lacking in their management scheme and how to remedy any shortcomings most efficiently. After only one year the benefits of conducting this program became evident to the surrounding plantation owners and managers at which time the size, scope, study area, and funding of the work was expanded. The project which began as a single 3-year study in 1992 continues to the present and is called the Albany Quail Project (AQP).

The AQP is a very unique program in several ways. It is a privately funded, university based research/outreach program where continued funding is closely tied to the quail hunting success experienced by the contributors. This type of accountability makes the AQP unique and accounts for the sharp focus of the work. Quail hunting success in this area is traditionally measured in covey finds/hour. This was the yardstick used to determine success of the Albany Quail Project from the beginning and continues today.

Another difference between the AQP and other university wildlife research projects is in the applied nature of the work and the speed at which it is transferred to the end users. Because of the close relationship between private landowners and researchers at the AQP, results from research are rapidly applied to management. This rapid integration of research and management was most productive when individuals involved with the hunting and land management activities of a given property were actively included in and consulted on all study directions, designs, and interpretations. This partnership gave ownership of the AQP to the non-university participants and served as a vital feed-back mechanism for university researchers. The other benefit of this cooperative effort was a significant decrease in the time from study completion to dissemination to the public. In many cases a manager or landowner would adopt management practices before completion of research without complete assurance of its efficacy. In comparison, university researchers usually require several years and levels of confidence before publishing results and making management recommendations.

A final diversion from typical university research projects is in how information from the project is distributed. The emphasis of the AQP is on the development of effective quail management techniques and disseminating these techniques to the public through newsletters, the popular press, and personal interactions rather than publishing in the scientific literature. While this did not necessarily gain favor with the university, it relates back to accountability and the source of continuing funding for the Project.

A principle component of the AQP was simply to monitor large numbers of radio-tagged bobwhite quail for long periods and then use those data to design field experiments to test dogmas and develop new management techniques to produce more quail and better hunting. The long-term nature and large sample sizes of the AQP have allowed management recommendations to be based on reliable evidence and continues to be successful as evidenced by the near 15-year existence of the AQP.

Herein, we highlight the results from the first 14 years of AQP. We present the 5 management items we deem critical for a successful quail management program in the 21st century in southern Georgia. Taken together these items provide the foundation for what we have termed Modern Quail Management (MQM), a formulae for producing quality bobwhite habitat and establishing a basis for future management decisions to increase bobwhite hunting success.

Study Area

Our study area is part of an 80,000 ha aggregate of large, privately owned properties in Baker and Dougherty Counties located in southwest Georgia surrounding the city of Albany. This land was purchased and developed as shooting properties mostly by wealthy northern industrialists beginning in the 1940s. The AQP began on one plantation of about 6,000 ha in size and has over time included 10 different properties that collectively represent over 42,000 ha (Sisson et al. 2009*b*). Approximately 80% of the study area consists of mature old-field pine woodlands interspersed with small (1-4 ha) and scattered fallow fields (20%). Land management activity through the 1970s and 1980s was typical of quail management in the region since the 1930s: a combination of timber thinning, prescribed burning, and agricultural plantings. Historically, quail populations over the study area fluctuated around 2.5 birds/ha (Simpson 1976). More detailed descriptions of our study area can be found in other AQP publications (Yates et al. 1995, Burger et al. 1998, Sisson et al. 2000*b*,*a*, Terhune et al. 2006, 2007).

Methods

The majority of our work focused on the use of radio-telemetry to monitor a wide array of bobwhite demographics, population ecology, behavior, habitat use, and hunting issues. The radio transmitter specifications and attachment procedures have been previously described and shown to produce accurate and reliable information for our study sites Sisson et al. (2009*a*,*b*), Terhune et al. (2007).

During the 15-year study many of the experiments and tests were evaluated by using hunting success (measured in coveys observed per hour) to measure the effect of a treatment. Since the objective of the quail plantations is to provide high quality quail hunting, the most direct method to evaluate any practice is to determine its effect on hunting success. This has been the traditional method used by owners and managers to evaluate population response to management programs in some cases for over 100 years, and has, more recently, been shown to correlate well with population density for these types of properties (Palmer et al. 2002). In addition to hunting success indices, we used surveys, censuses, radio-telemetry, and other data to determine a treatment's effect and explain why it occurred.

Many topics were addressed by the AQP from 1992 through 2005. Some of the most significant include the following:

Annual Survival and Causes of Mortality

To understand quail population dynamics it was important to document patterns in long-term survival and causes of mortality. Year-round monitoring of a large sample of radio-tagged birds on 2 study areas provided these data. Birds were trapped and radio-tagged at these study sites twice a year (Oct/Nov & March) and then monitored at least twice weekly throughout the year. Cause of mortality was recorded wherever possible by evidence at the kill site or condition of the radio as described by Curtis et al. (1988). Seasonal and annual survival estimates were obtained through the Kaplan-Meier method (Kaplan and Meier 1958, Pollock et al. 1989).

Brood Habitat Use

Very little was known about habitat requirements for quail broods in these environments when the study began, therefore this became one of our primary objectives early on. This information was produced by identifying habitat used by radio-tagged hens after their nests hatched until the young birds fledged.

Supplemental Feeding

Providing supplemental feed for birds has always been a traditional activity on quail plantations. This multiple-year study on intensively managed properties using wild, radio-tagged quail was designed to detect differences in bird survival, predation rates, reproductive output, and hunting efficiency on areas where supplemental feed was provided versus areas that were not. After this initial study was completed these same parameters were compared in two follow-up studies: 1) the effects of hunting season feeding versus year-round feeding and 2) the effects of spreading feed versus using stationary feeders.

Hardwood Clean-up

When the AQP work began in the early 1990s hardwood encroachment appeared to be a universal problem on all study areas. Using the same techniques and examining the same parameters as all the other studies from the AQP we compared responses of quail to "clean-up" operations (i.e. treatment) in pre versus post and treatment versus control (i.e. no hardwood removal) designs. The effect of "clean-up" on nest predator populations was documented as well.

Nest Predation

Numerous studies were conducted to better understand the relationship between quail and their predators. Radio-tagged quail were used to collect information about adult bird's predators, nest predators, and predator removal programs. We provided data from our study sites for additional research which involved developing and testing a "predator index" to determine the level at which predator populations should be managed and for a large-scale predator removal experiment. Most of our work involving predators was conducted in collaboration with other institutions and agencies such as Tall Timbers Research Station, University of Georgia, and USDA-APHIS Wildlife Services.

Results and Discussion

From 1992-2006 over 8,000 wild bobwhites were radio-tagged and monitored on 10 different properties in Georgia and Alabama by the field staff of the Albany Quail Project. One of the factors that became apparent early on in the investigations was that bobwhite survival was higher and distributed differently on these managed properties compared to other properties in the Southeast (Sisson et al. 2009*b*, Burger et al. 1998). In addition to higher annual survival (approx. 20%), mortality on our study areas was more evenly distributed throughout year giving relatively more breeding birds a chance to reproduce in spring and summer. The average overwinter survival for all our study sites combined was 54% and was shown to be the underlying theme for success on all the populations we have studied that are doing well, as well as the first sign of trouble for those that are not (Sisson et al. 2009*b*). We attribute this high survival to populations existing in large blocks of good habitat, a region of mild climate, conservative harvest pressure, management of mammalian predators, and supplemental feeding. In many other studies (Sisson et al. 2009*b*) the majority of annual mortality occurred during fall and over the winter leaving proportionally fewer breeders to produce young during the spring and summer.

Brood habitat was poorly understood at the beginning of the project with only a few small sample size studies having been conducted which produced apparently conflicting information. It became one of our initial objectives and a high priority to determine optimal brood raising conditions for our study sites.

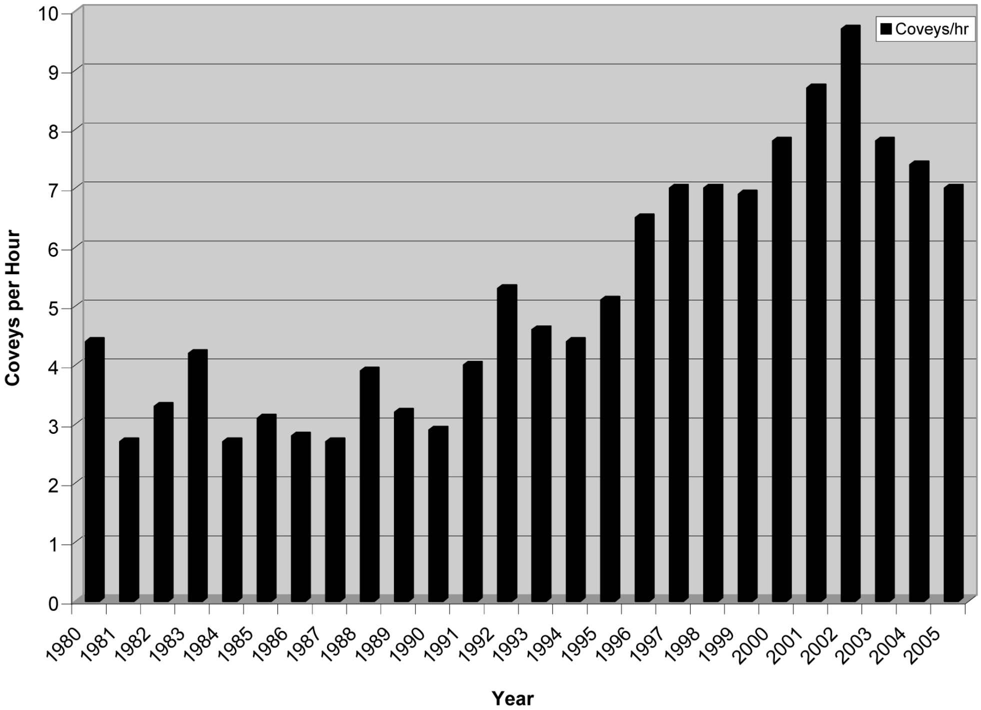

Figure 1: Hunting success (coveys/hour) on the primary study site of the Albany Quail Project in Baker County, GA prior to (1980 - 1996) and after (1997 - 2005) the implementation of Modern Quail Management techniques.

Monitoring radio-tagged hens with young broods revealed the importance of weedy fallow fields due to their cover conditions and high insect availability compared to other habitats (burned woods, agricultural fields, and "feed patches")(Yates et al. 1995). It was also determined that woodlands with a higher percentage of these types of fields produced consistently higher quality hunting with 30-35% of an area in fields being optimal (Michener et al. 2000). This resulted in a flurry of activity on local properties reclaiming old fields and creating new ones scattered across the landscape. Radio-telemetry monitoring of hundreds of broods since these early findings have re-enforced them and helped fine tune the management of thousands of acres of fallow "weed fields" now being maintained specifically for brood habitat. This was also our first indication that the traditional "bird patch" may have been ineffective in maintaining quail populations.

At the beginning of AQP in 1992 there was a long standing negative bias by wildlife professionals towards supplemental feeding of any kind (Frye 1954). However, this did not preclude it being a common practice for bobwhites on intensively managed properties throughout their range (Frye 1954, Guthery 1986, Simpson 1976, Brennan et al. 1994). It was under the influence of this negative bias that we began studying this topic in 1993 with this initial investigation providing some of the first evidence of the beneficial impacts of supplemental feeding for bobwhites (Sisson et al. 2000*a*). This study documented the potential of supplemental feeding to increase over winter survival, body condition, and reproductive output in the spring (Sisson et al. 2000*a*). Current large and longer-term studies at Tall Timbers Research Station are verifying these early results (W. H. Palmer, personal communication).

Moreover, follow-up investigations have revealed additional benefits to year-round feeding versus the traditional fall-spring feeding programs (AQP & TTRS, unpublished data). The next phase of this work was an examination of how supplemental feeding affected hunting success and hunting quality. A series of field experiments over a 4-year period revealed that coveys not being fed had a greater chance of being found by hunters (65%) than did fed coveys (50%)(Sisson 2005). This occurred for the same reasons that made unfed coveys in our earlier study more vulnerable to predators - increased daily movement and larger home range size (Sisson et al. 2000*a*). These findings contributed to the Georgia Department of Natural Resources policy that does not consider the hunting of quail that are being supplementally fed as "baiting". Broadcast spreading of supplemental feed year-round is now an integral part of most plantations management program.

These feeding studies also led to more questions concerning the long standing practice of planting quail "feed patches". During this same time period we conducted an experiment systematically eliminating feed patches on one of our study areas without negatively impacting the quail population (AQP, unpublished data), as was similarly reported on by Brennan et al. (2000). This in conjunction with the lack of use by broods, as mentioned above, contributed to the widespread demise of this long standing quail management tradition on many sites.

Hardwood encroachment into old-field pine uplands over time became one of the biggest management challenges of properties in the area. Even with frequent burning regimes, abundant mid- and overstory hardwoods had over time encroached to the point where ground cover growth was being negatively impacted. As this problem began to be dealt with in the Albany area, we initiated a long term telemetry project to monitor bobwhite demographics, population density, and predator abundance before and after this intensive mechanical hardwood removal and compared to other untreated (i.e. hardwoods not removed) sites. The results of this study illustrated dramatic improvements in bobwhite survival and reproductive success following a mechanical hardwood "clean-up" operation, as well as dramatic reduction in predation stemming from the removal of predator habitat (Sisson et al. 2002). Quail populations on these sites soon reached an all time high and remain high and stable today. Hardwood removal is not a novel idea, previously being described by Rosene (1969) but had not received much attention recently. However, in response to illustrating the benefits anew, this type of work has been conducted over tens of thousands of hectares in the region during the last decade.

The management of nest predators has long been a controversial subject with little research specific to bobwhites available to make management decisions. Our telemetry studies along with those of our colleagues were raising concerns about the level of nest predation occurring on sites in the southeast. In addition, new technology was documenting precisely what the main nest predators were (Staller et al. 2005). The lack of comprehensive knowledge on management of quail nest predators led to a series of research experiments addressing the subject. The initial work was a Quail Unlimited funded collaborative project (AQP, TTRS, MSU) looking at nest predator abundance region wide that documented a strong correlation between predator abundance and reproductive success (Palmer et al. 2001). This was followed up by an on-going collaborative project (AQP, TTRS, UGA, USDA-WS) looking at experimental nest predator removal in a cross-over design and replicated across study sites. These studies are demonstrating the positive impacts of nest predator management under some circumstances (Carroll et al. 2005). In response to the results of these studies and interest from landowners, the Georgia Department of Natural Resources now has a permitting system that allows landowners with a management program in place for bobwhites to trap nest predators year-round under special permit. This program has been a huge success and is now standard procedure on most of the intensively managed properties in the region.

The result of the above described public/private collaboration has been a change in management

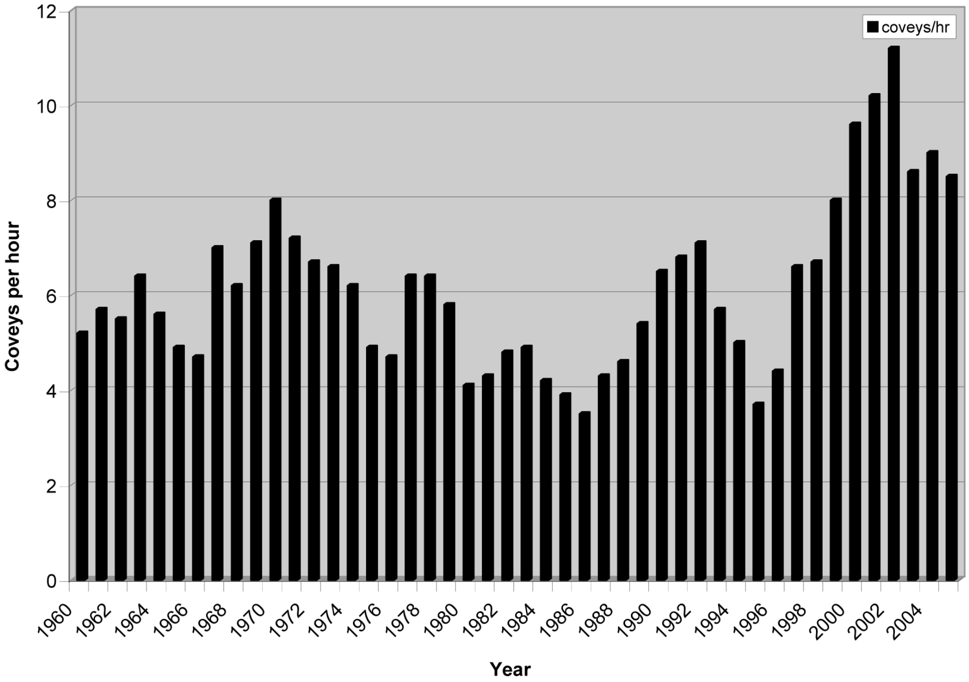

Figure 2: Hunting success (coveys/hour) on the secondary study site of the Albany Quail Project in Dougherty and Baker Counties, GA prior to (1960 - 1996) and after (1997-2005) the implementation of Modern Quail Management techniques.

style across a large acreage to one based on science creating the concept of modern quail management (MQM) techniques. This has taken plantation style quail management above and beyond the standard techniques of thinning timber and burning the woods by adding more aggressive techniques. The cornerstones of this new management philosophy have been hardwood clean-up, brood habitat management, management of nest predators, and year-round supplemental feeding.

Bobwhite population densities and hunting success during the last decade on these properties implementing MQM are now higher and more stable than any previous period in their history. On our primary study site from 1980 to 1996, prior to their initiation of MQM, the average number of coveys seen per hour was 3.9 (SE = 0.265) with a range of 2.7-6.5 (CV = 0.28). Following the implementation of MQM, from 1997 to 2005, average number of coveys seen per hour almost doubled to 7.7 (SE = 0.317) with a range of 6.9-9.7 (CV = 0.12, Figure 1). Population density estimates on the same property during the same time frames showed similar increases from about 2.5 birds/ha to at or near 5.0 birds/ha. Our secondary study area had a covey per hour average of 5.5 (SE = 0.188) during the period from 1960-1996 with a range of 3.7-8 (CV = 0.21). Since implementation of MQM this property has averaged 8.7 coveys per hour (SE = 0.503) with a range of 6.6-11.2 (CV = 0.17, Figure 2), a testament to the efficacy of tried and applied research and management.

Management Implications

Modern quail management is a much more intensive form of quail management than was practiced prior to the time the Albany Quail Project began. Traditional quail management techniques were developed in conjunction with land use practices that existed from the 1930s through the late 1960s. Under traditional management programs, bobwhite densities fluctuated around 2.5 birds/ha and were largely a byproduct of other land practices (Stoddard 1931, Rosene 1969, Simpson 1976, Palmer et al. 2002). Today's properties practicing MQM now have populations fluctuating around 5.0 birds/ha and higher.

Successes from the research and outreach efforts of the AQP helped created optimism in the quail community which has led to increased management intensity on many properties. Modern quail management has spread and succeeded on traditional quail management areas first developed in the early 1900s in Alabama, throughout Georgia, the Carolinas, into Virginia and elsewhere. In addition, the AQP and its approach have been involved in developing successful new quail properties on southeastern land holdings which were never previously dedicated to quail production. Even with the high cost of renovations, annual input, and foregone alternative revenue, properties with large populations of wild birds and quality quail hunting are in high demand but extremely limited in supply. Surveys of firms involved in brokerage of such properties indicate that value added to a hunting property managed using MQM techniques approaches $1,000 per acre more than a similar property managed in the traditional manner.

Since its conception, the AQP has helped increase knowledge of management for quail, do away with some dogmas, and break new ground on issues such as feeding, brood habitat management, and the benefits of predator management. Its research and successes helped influence revisions in state wildlife management policy and regulations to now allow supplemental feeding during hunting season and permitted year round trapping of mammal nest predators. The one fact that those involved with the project always keep in the forefront of their thinking is that the AQP began because people involved with quail management realized that the world in which their quail populations existed had changed but management had not. Modern quail management techniques are extremely successful today however, "Modern" is a relative concept, which eventually becomes "Traditional". Continued research and application of those results are needed to ensure high density bobwhite populations and quality hunting over the long term.

Acknowledgments

We have had the distinct pleasure over the years of working on some of the finest properties in the country as well as benefiting from the knowledge of some of the most skilled quail managers in existence. It has been a rare treat as researchers to see results of our work immediately put into action and produce results. As always, we thank the landowners and others who have supported this effort and all the dedicated field staff that gathered all the data and made us look good.

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Summary of Georgia's Bobwhite Quail Initiative 2000-2005

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Georgia's Bobwhite Quail Initiative (BQI) has been a proactive effort to restore and improve habitat for northern bobwhite (*Colinus virginianus*; hereafter bobwhite) on private lands across a 15 county area of Georgia's Upper Coastal Plain. Secondary objectives included improving habitat for certain songbirds, quail hunting and wildlife viewing. The BQI provided landowners/managers (Cooperators) with technical assistance, and through a competitive process, financial incentives for bobwhite habitat management. The Georgia General Assembly and Department of Natural Resources (DNR) Board initiated BQI in 1998 in response to hunter/constituent concern over declining bobwhite populations; and the Georgia DNR Wildlife Resources Division began implementation in 1999. Funding for BQI was provided through state appropriations, including funds generated through the sale of BQI vehicle license plates. Habitat incentive payments and practice options were modified gradually to increase Cooperator participation and to better integrate bobwhite management with commercial agriculture and forestry. Research and monitoring indicated positive impacts of habitat practices on bobwhites and certain songbirds. The BQI generated many additional benefits including leveraged funding for management and research; youth quota quail hunts; and increased educational outreach regarding the bobwhite decline and effective restoration techniques. A Cooperator survey indicated high customer satisfaction and a strong perception that BQI practices have improved bobwhite and songbird populations, as well as the environmental condition on Cooperator farms. The BQI showed that: 1) bobwhite numbers can be increased on working farm and forestlands, and 2) adequate levels of economic incentives and qualified technical staff are essential for success.

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Key words: *Colinus virginianus*, farmland, forestland, Georgia, incentives, landowners, northern bobwhite, Upper Coastal Plain

Introduction

The northern bobwhite (*Colinus virginianus*, hereafter bobwhite) and bobwhite hunting are prominent in Georgia's wildlife heritage. Consequently, in 1970 the Georgia General Assembly (Assembly) designated the bobwhite as the official state game bird. However, bobwhite populations in Georgia and across the southeastern United States have experienced severe long-term declines (Sauer et al. 2005) primarily as the result of widespread changes in land use (Klimstra 1982, Brennan 1991, Burger 2002). The Northern Bobwhite Conservation Initiative (NBCI) identified nesting cover and brood range as the landscape habitat components most often limiting bobwhite abundance, and recommended restoration of these habitats on agricultural and forestlands as a priority for bobwhite population recovery (Dimmick et al. 2002).

In Georgia the bobwhite decline has been cause for concern ecologically, economically and recreationally (Thackston and Whitney 2001, Burger et al. 1999). In 1964 there were 127,000 quail hunters who comprised 47% of the state's licensed resident hunters and harvested an estimated 3,365,000 quail (Georgia Game and Fish Commission 1965). In 2002, the number of quail hunters dropped to 29,858, and comprised only 12% of the licensed resident hunters; these hunters harvested an estimated 541,922 quail, of which approximately 68.5% were reported as pen-reared birds (Nicholson 2003). In much of Geor-

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gia, bobwhite densities have fallen below the level needed to attract and maintain hunter interest, and in some areas, particularly in the northern half of the state, viable bobwhite populations no longer exist.

In 1995, the Georgia Department of Natural Resources (DNR) Wildlife Resources Division (WRD) began a Private Lands Initiative (PLI) directed at improving the composition and delivery of Farm Bill conservation programs and practices on privately owned agricultural and forested lands. The PLI emphasized enhancing habitat for bobwhites and other grassland dependent species. In 1998, the Chairman of the Georgia House of Representatives, Game, Fish and Parks Committee contacted WRD administrators on behalf of concerned constituents requesting action to restore bobwhite populations, primarily for the purpose of improving quail hunting. This ″grass roots″ concern worked in synergy with the PLI to facilitate increased interaction of WRD with the Assembly and DNR Board Members (Board) to address the bobwhite decline. This interaction led to the funding and development of Georgia's Bobwhite Quail Initiative (BQI), a pilot program to restore early successional habitat on commercial row crop agricultural fields and associated forestlands. The BQI was Georgia's first state funded private lands wildlife habitat incentive program. This paper summarizes and discusses the BQI developmental process, implementation, and management implications.

Developmental Process

The WRD initiated the developmental process by assigning a PLI staff biologist to oversee and write the BQI plan. Soon thereafter a multi-organizational team of biologists and administrators was assembled to provide input in establishing plan goals, objectives and other procedural components. This proved beneficial for: (1) garnering ideas and information from varying perspectives, (2) accruing buy-in and support from other natural resource organizations, and (3) securing outside credibility or validation for WRD recommendations.

A series of team meetings were conducted where plan components were identified, discussed and refined. A proposed plan was developed, Georgia's Bobwhite Quail Initiative (Georgia DNR 1999), that covered the: (1) bobwhite decline causative factors, (2) goals and objectives, (3) monitoring, (4) personnel and equipment, and (5) proposed budget. The primary goal was to restore bobwhite habitat on commercial row crop fields. Secondary goals included enhancing habitat conditions for early succession dependent wildlife, particularly certain songbirds that were in serious decline, and increasing opportunities for wildlife viewing.

The final proposal included three spatially explicit restoration alternatives and associated budgets: (1) statewide for 10 years at a total cost of $40 million, (2) 47 counties in Georgia's Upper Coastal Plain Physiographic Province (UCP) at an annual cost of $2.3 million, and (3) 12 - 15 counties in the UCP at an annual cost of $1.2 million. After subsequent meetings between WRD, Assembly and Board members, the final plan included 14 counties in the UCP and an annual budget of $939,000. Counties were chosen through a bio-political selection process designed to maximize the probability of successful implementation. Generally, chosen counties were in the UCP and were predominately agricultural with commercial row crops comprising more than 40% of this acreage. Additionally, in an attempt to create a viable habitat matrix at the landscape level, the program was focused on a limited number of counties in close proximity to each other. During 2000-2003 BQI county numbers changed due to budget fluctuations. County numbers increased to 17 in 2001 as appropriations increased and then were reduced to 15 counties in 2003 when state budget reductions occurred (Figure 1).

In 1999 WRD administrators, the Board and supporting Assembly members successfully moved the proposal through the state legislature for final approval and funding. In large part this effort was successful because supporting Assembly members served on, and chaired, key committees as follows: 6 on House Appropriations, 2 on House Game, Fish and Parks, 2 on Natural Resources and Environment, 4 on Senate Appropriations, and 1 on Senate Natural Resources.

Implementation

The primary BQI implementation components included: (1) securing funding, (2) delivering the program, (3) initiating program promotion and educational outreach, (4) developing habitat incentive scoring guidelines and a competitive ranking process, (5) enrolling landowners/managers (Cooperators) for habitat incentives, and (6) conducting monitoring, research and surveys.

During 1999-2001 program funding was derived solely from appropriations of tax revenue from Georgia's General Fund. However, during 2001 the Assembly developed and passed legislation creating a BQI automobile license plate (tag). Prior to finalizing the tag design, WRD surveyed a diversity of public groups relative to their preferences of several different tag prototypes. The most popular design included a large whitetail (*Odocoileus virginianus*) buck and a covey of bobwhites. During the period 2001 - March 2006, 336,265 BQI tags were sold, which generated $5,777,642 net revenue. Since this exceeded the BQI budget, the overage was used to reimburse the State General Fund for the years that BQI operated without a dedicated funding source.

The WRD contracted with the Georgia Soil and Water Conservation Commission (SWCC) for the distribution of Cooperator habitat incentives. This enabled WRD to carry BQI funding across fiscal years without reverting revenue back to the General Fund, which was key to having guaranteed funding for 3-yr Cooperator contract cycles.

In addition to annual appropriations, BQI funding has been used as a match to secure over $100,000 in grants and donations. It has also been used as a match for a contribution agreement with the Natural Resources Conservation Service (NRCS) to fund a biologist position to deliver Farm Bill programs.

In 1999, a BQI program coordinator, 2 secretaries, 5 field biologists and a public relations specialist were hired to develop and deliver the operational aspects of the program. Four field offices were established and the BQI focus counties were split into three administrative focal areas (Figure 1).

An interdisciplinary task force of wildlife biologists, foresters, agricultural administrators and wildlife researchers was formed to assist with developing specific habitat practice guidelines and incentive payments. These scoring guidelines were used to define and assign point values to habitat practices, set payment rates, and competitively score, rank and fund habitat proposals (Table 1, Appendix A). Through a feedback process these practice guidelines and incentive rates were temporally modified to increase participation, better integrate bobwhite management with Cooperator objectives and optimize the cost benefit ratio. A database was established for tracking Cooperator participation, practice hectares, and incentive allocations.

Cooperator participation in BQI was strictly voluntary. Prospective Cooperators contacted BQI biologists for technical assistance and/or potential enrollment for habitat incentives. Biologists worked intensively with Cooperators to develop detailed technical assistance plans, and/or habitat incentive proposals to integrate bobwhite management with other resource objectives. Cooperators then decided whether or not to submit their proposals for ranking. At the end of each enrollment period, incentive proposals were scored for habitat quality and competitively ranked for funding. Cooperator habitat proposals had to exceed, and if enrolled be maintained above, a quality threshold to enter and remain in the program. Enrolled Cooperators signed 3-year contracts with WRD, which included detailed prescriptions for habitat practice establishment and maintenance. Biologists flagged or marked habitat practices on enrolled lands, Cooperators implemented management as prescribed in the contract and biologists conducted annual contract compliance evaluations. If habitats were in compliance then contracts were approved and SWCC disbursed incentives. If not, depending on the severity of non-compliance, Cooperator contracts were either voided with no payment, or they were amended to provide partial payment. In either case, biologists encouraged Co-

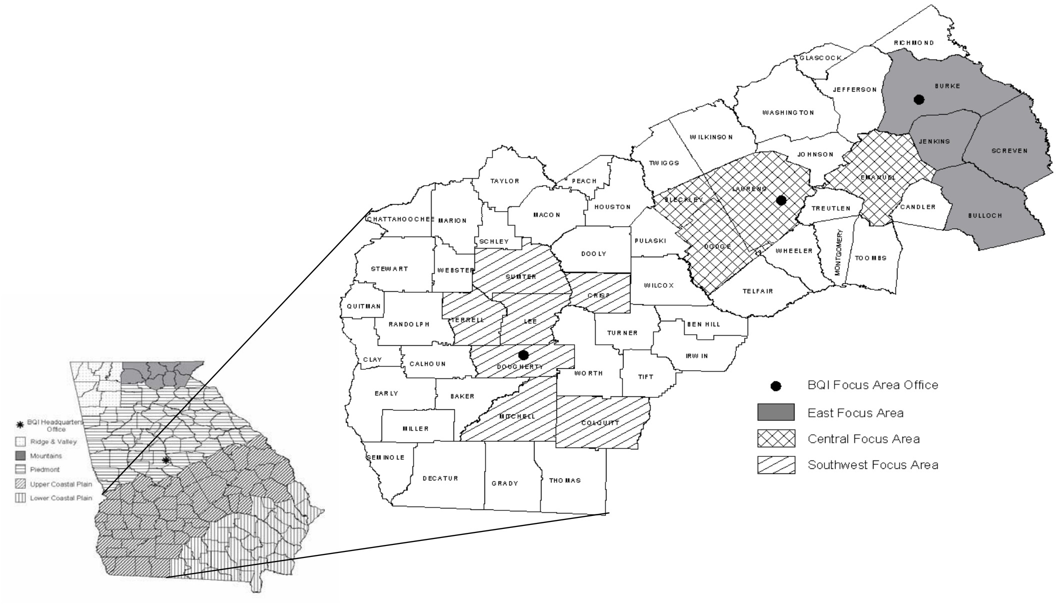

Figure 1: The BQI focus areas in the Georgia Upper Coastal Plain 2000 - 2005.

operators to remain in the program and strive to fully meet practice requirements the following year.

Monitoring, research and survey projects were conducted under contract with the University of Georgia D.B. Warnell School of Forest Resources (UGA) and by BQI biologists. Monitoring was conducted by UGA in 1999-2001 for bobwhites (Hamrick 2002) and songbirds (Hamrick et al. 2001) and then, due to state budget reductions, by BQI biologists for bobwhites only during 2003-2005. Research projects were conducted to determine: (1) the impacts of bermudagrass (*Cynodon dactlylon*) on bobwhite chicks (Burkhart 2004), (2) efficacy of various herbicides for controlling bermudagrass (Burkhart 2004, Bond et al. 2005), and (3) bobwhite utilization of BQI habitats (Cook 2004). Additionally, in 2004 BQI Cooperators were telephone surveyed relative to their opinions on BQI and its impact on bobwhites, other wildlife and quail hunting on their land.

Discussion

Education and Outreach

Education and outreach have been identified as important actions relative to effecting positive habitat change for bobwhites on private lands (Brennan 1991, Capel et al. 1996). Outreach efforts were viewed as essential to BQI for informing the public about the: (1) reasons for, and solutions to, the bobwhite decline, (2) multiple resource benefits of BQI practices, and (3) availability of technical assistance and habitat practice incentives. Outreach efforts were varied and generated keen public awareness and interest in BQI (Table 2), which became evident as the number of public inquiries and political support for the program grew. Empirically, the greatest Cooperator response to outreach efforts came from: (1) BQI articles published in the Georgia Farmers Market Bulletin (a biweekly publication by the Georgia Department of Agriculture), (2) BQI town hall meetings (conducted in 12 of the 15 the BQI counties during the first 3 years of the program), (3) BQI Newsletter publication (published 9 times during 2000-2005 and distributed free of charge to more than 1,500 people), (4) enrollment of certain

Table 1: Northern bobwhite habitat practices, point values and incentive rates in Georgia's BQI 2005 (see Appendix A for additional details).

	Habitat Practice Point Values		**Practice Incentive Annual Rate Per Hectare (ha)**	
Habitat Practice	Per Practice	Maximum	Dry	Irrigated
Field borders	25	150	$148.26	$296.52
Hedgerows	10	40	$148.26	$296.52
Filter strips	10	40	$148.26	$296.52
Center pivot corners	10	40	$148.26	NA
Fallow patches	4.9/ha	20	$148.26	NA
Conservation tillage	10	20	$37.07	NA
Pine forest thinning	2.5/ha	50	$37.07	NA
Pine forest openings	2.5/ha	50	$148.26	NA
Pine forest linear practices	2.5/ha	20	$148.26	NA
Pine forest burning,disking, herbicide	12.4/ha	100	$12.36	NA
Crop field bonus points	5 to 15	20	NA	NA
Pine forest bonus points	5 to 30	40	NA	NA
Habitat connectivity	5	20	NA	NA
Reduction in funding	5 to 20	20	NA	NA

landowners who were considered to be "leading farmers" within a particular county, (5) flyers with program and contact information placed at county FSA/NRCS offices, and (6) mailings to landowners whose names were on mailing lists obtained from the NRCS district offices. Also of importance were the many detailed management notes and technical publications that were developed for specific habitat practices and resource concerns, and were frequently used in technical assistance and educational outreach. An informational video was produced as a training and outreach tool but had limited utility in promoting the program or increasing Cooperator enrollment.

Indirectly related to education and outreach was the hosting of quota youth quail hunts on BQI enrolled farms. Cooperators voluntarily hosted these hunts. The BQI biologists assisted with conducting hunts, but WRD assumed no liability. These hunts were labor intensive but successfully introduced a limited number of youth to wild quail hunting. During 2003-2005 a total of 19 hunts were conducted on 12 BQI enrolled farms, hosting 38 youth/adult pairs, who hunted 118 hours, found 59 coveys and harvested 19 quail. Most of the participating youth had never quail hunted and both youth and adults provided positive feedback.

Technical Assistance

The demand for BQI technical assistance was high. During 2000-2005, BQI biologists provided detailed management recommendations to 815 landowners on 168,227 hectares. A major benefit of BQI was having full-time biologists that worked specifically on the restoration of bobwhite on private lands. This enabled follow up site visits and fine tuning of management practices. This service intensity would not have been available from WRD regional biologists who were multi-tasked over large geographic areas comprised of both public and private lands. The educational aspect of the technical assistance program was another apparent, albeit difficult to measure, program attribute. Interacting in-

Table 2: Summary of BQI public information and education effort 2000 - 2005.

Activity	2000	2001	2002	2003	2004	2005	Totals
Programs/presentations	14	6	18	17	30	25	110
Field day presentations	2	8	12	22	11	1	56
Town hall meetings	6	7	3	0	0	0	16
Presentation attendees	751	888	2476	2738	1650	1476	9979
Professional articles/abstracts	0	1	6	2	4	0	13
Popular articles	2	4	11	8	19	9	53
BQI newsletters	2	2	2	1	1	1	9
Television spots	1	4	2	4	0	0	11
Display booth man-days	5	10	7	8	7	2	39

dividually with Cooperators in the field provided opportunities to impart information, and influence opinions and decision-making, to an extent that may not be accomplished through mass media.

Enrollment

Demand for BQI habitat practice incentives was initially low but increased through time as practice options and payment rates were increased. In 2000, the habitat practice incentive payment was $74.13 per hectare per year for linear practices on crop fields for both dry and irrigated lands. At this incentive rate, the number of Cooperator proposals did not meet the available BQI funding. In response to this lack of enrollment, incentive rates were quickly modified to $98.84 per hectare for dry land linear practices and $296.52 per hectare for irrigated linear practices. Conservation Reserve Program Longleaf Pine Conservation Priority Area enrolled fields were made eligible for the BQI field border practice. Cooperator proposals increased but still did not exceed available funding. In 2001, the BQI dry land incentive rate for linear practices was increased to $148.26 per hectare per year, the conservation tillage practice payment was increased from $2.47 to $37.07 per hectare per year, and the maximum payment cap was increased from $10,000 to $15,000 per Cooperator per contract. Additionally, a suite of forest management practices was added. With these changes, proposal numbers increased to the point of exceeding available funding in 2003.

During 2000-2005 the U. S. Department of Agriculture Farm Service Agency dry land and irrigated cash rental rates for the BQI counties averaged $111.20 and $370.65 per hectare, respectively (C. Weaver, U.S. Department of Agriculture, personal communication). Cooperators were unwilling to enroll field acres into BQI until the habitat incentive exceeded the crop rental rate; hence only 9% of the enrolled crop field acres were irrigated. Cooperators could choose from a variety of eligible management practices (Table 1, Appendix A) but the following 5 were used on over 80% of the total hectares enrolled during 2000-2005: (1) conservation tillage (other BQI practices required) - 26.3%, (2) 9.1 meter wide field borders - 18.5%, (3) managed pine plantations - 16.4%, (4) fallow managed patches 0.4 to 4.0 ha in size - 12.9%, and (5) 18.2 meter wide field borders - 7.4%.

Habitat enhancement peaked in 2003 at 3,274 direct practice hectares and 8,381 impact hectares (Table 3) where direct practice hectares were those specifically manipulated with the BQI practice and impact hectares were defined as the total area within a crop field or forest stand treated with BQI practices. This represents an annual cost of about $71.66 per actual practice hectare and $27.18 per impact hectare. In 2003, there were 132 Cooperators enrolled for habitat incentives and impact hectares were distributed across 94 pine stands and 289 crop fields, including 663 kilometers of field borders, hedgerows and filter strips. Beginning in 2003, lack of funding became the limiting factor for enrolling Cooperators and positively impacting bobwhite habitat.

Compliance

Each year after crops were planted and/or harvested (May-September), compliance evaluations were conducted by BQI biologists on all of the enrolled practice hectares. Practices were categorized as: (1) full compliance - at least 80% of the habitat practices were properly established and maintained, (2) partial compliance - less than 80% of the practices were properly established and maintained but the total habitat score remained above the minimum quality threshold necessary for entry into BQI, and (3) non-compliance - less than 80% of the habitat practices were established and maintained and the habitat score was below the minimum BQI threshold. Across all years landowners averaged 72% full compliance, 23% partial compliance and 3% non-compliance. Compliance remained high throughout the program and non-compliance decreased as biologists worked with landowners to explain practices and resolve issues. Again, this points to the value of having sufficient numbers of technical staff to work closely with Cooperators.

Cooperator Survey

A 2004 telephone survey of 102 BQI Cooperators (Appendix B) showed: (1) Cooperator satisfaction with BQI was high; (2) prior to BQI most Cooperators were not implementing BQI practices on their lands and the most common pre-BQI practice for bobwhites was planting food plots (48%); (3) post BQI the most common practice implemented was weedy field borders (96%) followed by weedy hedgerows (82%) and weedy field corners (81%); and (4) most Cooperators felt BQI had improved their land's: environmental condition (91%), quail population (81%), quail hunting (79%), and songbirds and other wildlife (82%). However, only 24% said they would have implemented BQI practices without the provision of economic incentives.

Research

BOBWHITE AND SONGBIRD MONITORING - The habitat practices promoted and funded in BQI have been shown or recommended to impact positively bobwhites (Stoddard 1931, Rosene 1969, Minser and Dimmick 1988, Palmer et al. 2001). However, to assess and validate BQI practice impacts, monitoring was conducted on BQI treatment and control fields for bobwhites during 1999-2001 and 2003-2005; and for songbirds during 2000-2001. Bobwhites were surveyed during 1999-2001 with fall covey counts (Carroll 2000, Hamrick 2002). Carroll (2000) reported a pre-treatment average of 2.22 coveys per property across 12 treatment and 18 control fields and felt enough bobwhites were present to respond to habitat enhancements. Hamrick (2002) reported increasing trends in bobwhite numbers on BQI treatment fields post treatment and declining numbers on control fields, and concluded that BQI practices were positively impacting bobwhite populations. Winter songbird populations were also affected; a 30% increase was detected for nine sparrow species with three of these species, Le Conte's (*Ammoddramus leconteii*), grasshopper (*Ammoddramus savannarum*), and white-crowned (*Zonotrichia leucophrys*) occurring only on the first year post treatment (Hamrick et al. 2001). State budget reductions in 2001 forced WRD administrators to choose between reducing BQI habitat incentives and reducing monitoring. The decision was made to curtail intensive monitoring in favor of maximizing habitat establishment and maintenance.

In 2003, BQI biologists began recording incidental sightings and calling of bobwhites while conducting habitat compliance evaluations on treatment fields and then similarly walked and surveyed a random sample of control fields. This technique was not standardized over time or area, and did not provide an estimation of bobwhite density. However, analyses of variances were used to test for differences

Table 3: BQI Cooperator and habitat enrollment and incentive allocation by contract period 2000 - 2005.

BQI Contract Period	Number BQI Counties	Number Cooperators	Number Crop Fields	Longleaf CPA Stands	Number Pine Stands	Direct Practice Hectares[a]	Impact Hectares[b]	Incentives Allocated
2000	14	24	69	1	0	174	1,225	$17,093.40
2000 - 01	14	83	136	58	0	778	4,642	$64,030.20
2000 - 02	17	93	176	57	1	1,237	5,555	$78,355.90
2001 - 03	17	132	289	70	24	3,275	8,381	$233,827.00
2002 - 04	17	108	253	24	24	3,044	6,524	$221,465.40
2003 - 05	15	137	292	38	23	3,169	7,306	$258,544.70

[a] Direct practice hectares are those actually enrolled in BQI.

[b] Impact hectares are those included in the field on which BQI practices are implemented.

in bobwhite occurrence within treatments and controls across years (Sokal and Rohlf 1981). Tests were conducted using the Data Analysis Toolpak in Microsoft Excel©. No significant differences were detected across years for treatments (2003 $n = 252$, 2004 $n = 169$, 2005 $n = 95$; F = 2.115, df = 2, $P = 0.122$) or controls ($n = 2003$ $n = 39$, 2004 $n = 26$, 2005 $n = 28$; $F = 1.356$, df = 2, $P = 0.263$). Therefore, data were pooled across years and tested with an analysis of variance between treatments and controls (Sokal and Rohlf 1981). Treatments averaged 2.02 quail per field (SE = 0.16) and were significantly greater than the controls that averaged 0.92 quail per field (SE = 0.22; treatment $n = 516$, control $n = 93$; $F = 8.008$, df = 1, $P = 0.005$).

BOBWHITE HABITAT USE - Cook (2004) examined bobwhite breeding season dispersal, habitat use and survival in relation to agricultural lands with BQI habitat practices. He concluded that BQI habitats were utilized by adult birds and extensively by broods, and had a positive effect on bobwhite breeding season survival. Additionally, he concluded that closed canopy pine stands negatively impacted bobwhite winter survival and recommended thinning and burning these stands as a high priority for bobwhite restoration.

FIELD MARGIN VEGETATION - Burkhart (2004) examined vegetation response in BQI field margin habitats, the potential negative impacts of bermudagrass invasion and the control efficacy of certain grass selective herbicides. He found a positive response of desirable vegetation in both species composition and structure on BQI habitats during the first and second post treatment growing seasons. He also documented significant invasion of bermudagrass into fallowed BQI habitats and concluded that at high density it reduced bobwhite chick mobility, increased heat stress and potentially reduced chick survival. He evaluated a single application of 2 grass selective herbicides, Fusion (fluazifop p-butyl, fenoxaprop-p-ethyl) and Select 2EC (clethodim) and found them to be ineffective for significant bermudagrass control and bobwhite habitat enhancement in BQI field borders. Bermudagrass invasion was judged to be a significant problem across more than 50% of BQI field margin habitats. Therefore, BQI biologists conducted additional research to determine effective control techniques. Bond et al. (2005) determined spring burning followed by a summer application of Chopper (Imazapyr) at 0.84 kg ai/ha was the most effective technique to reduce bermudagrass density and enhance bobwhite habitat. Subsequent to this research, bermudagrass control was included as an optional practice in BQI.

FUTURE RESEARCH - In addition to site-specific habitat quality, landscape context (Roseberry and Sudkamp 1998) and thresholds of usable space (Guthery et al. 2000) have been identified as important factors for bobwhite population restoration and long-term viability. The North Carolina Cooperative Upland Habitat Restoration Program, a state implemented bobwhite and early successional habitat initiative, sought to address this issue by forming landowner cooperatives with $\geq$ 2,025 ha of potential habitat (Cobb et al. 2002). Likewise, habitat fragmentation was recognized as a serious impediment to bobwhite restoration during the BQI developmental phase. To address this concern, BQI was focused in only 15 of Georgia's 159 counties. Expectations were that enough land would be enrolled in BQI to positively impact habitat conditions at the landscape scale, and by default create habitat units of sufficient size to support metapopulations that would be enlarged through increasing landowner participation. This appears to have occurred in some areas where large numbers of BQI crop fields and forestlands are in close proximity. However, there are still numerous BQI treatment sites isolated in landscapes of poor habitat quality, and there are large geographic areas within BQI counties that are completely void of bobwhite habitat.

Hamrick (2002) and Cook (2004) speculated that the magnitude of bobwhite population response to BQI treatments was influenced largely by adjacent habitats and recommended additional research on this topic to assess bobwhite population response to BQI treatments. Habitat modeling of the BQI landscape variables relative to bobwhite population density is needed to set quantitative habitat objectives and facilitate identifying sub-county geographic focus areas where higher habitat incentive rates could be justified. Through this effort these habitat islands and metapopulations could be expanded spatially. Currently, an adaptive resource management analysis of BQI treatments and landscape context is in progress at UGA with the objective of providing results that can be used more efficiently to focus BQI effort and habitat incentives.

Management Implications

Dimmick et al. (2002) recommended enhancing habitat on 6% to 7% of farm, forest and range land to achieve NBCI restoration objectives. Managing 7% of the cropland and pine forest hectares found in the BQI focus counties would have entailed directly treating 21,534 hectares of cropland and 33,489 hectares of pine forests (Thompson 1998, United States Department of Agriculture 1999). During 2001-2003, the peak BQI enrollment period, habitat practices directly impacted 2,350 hectares of cropland and 925 hectares of pine forest. This represented an 11% and 3% attainment of the cropland and pine forest goals, respectively. To fully achieve NBCI goals with BQI funding alone would have cost an additional $3.5 million per year. However, during 2005, Farm Bill Conservation program funding in Georgia for the Conservation Reserve Program, Wildlife Habitat Incentives Program and upland management practices of the Environmental Quality Incentives Program exceeded $15 million (C. Weaver and K. Wooster, U.S. Department of Agriculture, personal communications). All of these programs have a wildlife objective and theoretically a large portion of these funds could have been directed toward early succession habitat management in the BQI focus counties.

Georgia's BQI has shown that bobwhite populations can be increased across working farm and forestlands through focused funding and habitat improvement. In addition to increased habitat for bobwhites and other early succession dependent wildlife, BQI has served to: leverage additional funding for monitoring, research and habitat incentives; increase wildlife conservation education and outreach; and provide public access to private lands for quail hunting. The program has been popular with the public, private landowners and has received strong political support. It has shown that sufficient funding for habitat incentives and adequate numbers of technical staff are critical for enhancing bobwhite habitat on private lands. State initiatives are vitally important for contributing to the attainment of NBCI goals and objectives, but additional sources of funding will be required and Farm Bill conservation programs offer the greatest potential for making bobwhite recovery a reality. Public awareness, financial support and contributions from private conservation groups like Quail Unlimited® and Quail Forever® also are important components for success. The bobwhite decline can be abated, but the question is do we as a profession, and as a society, have the collective will to allocate the resources necessary to solve the problem?

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Appendix A. Detailed information on implementation of Georgia's BQI program including habitat eligibility, funding and scoring guidelines.

1) ELIGIBILITY:

   a) Properties must be located in one of the three BQI focus areas

   b) Commercial shooting preserves are not eligible for incentives

   c) Minimum property size for enrollment is 20.2 contiguous hectares

   d) Incentives will not be provided for previously-established habitat (See section 5o for protocol on the determination of previously established habitat)

   e) Properties eligible for financial incentives must include commercial row crop agriculture, fields enrolled in the Conservation Reserve Program Longleaf Pine Conservation Priority Area (LLCPA), or pine forests immediately adjacent to either row crop agriculture or LLCPA fields enrolled in BQI.

   f) After initial enrollment and during the remainder of the BQI Contract, the area must be maintained as either commercial row crops, land enrolled in the LLCPA, or in a cover conducive to quail management, as determined by the Wildlife Resources Division (WRD) biologist.

   g) Whole fields may be fallowed (i.e. cease to be commercially cropped) after initial enrollment. However, the fallowed hectares not under BQI contract must be managed at least once during the three-year contract period by winter disking, planting to a cover approved by the BQI biologist, or commercial cropping.

   h) Properties enrolled in the LLCPA will be scored equivalent to commercial crop fields and will not be eligible for Pine Forest Management Practices unless deemed appropriate by the WRD biologist

   i) Sites predominated (>51%) by exotic grasses (Bermuda, Bahia, Fescue) will be ineligible for enrollment until the grasses have been chemically treated, as recommended by the WRD biologist.

   j) To be re-enrolled in BQI properties must maintain eligibility.

2) ENROLLMENT:

   a) Sign-up is continuous, although ranking and enrollment will be based on the funding cycle

   b) Cooperators can apply by obtaining an application from the WRD, Soil and Water Conservation Commission, Natural Resources Conservation Service, Georgia Forestry Commission, or Farm Service Agency

   c) Proof of ownership or proof of owner agreement to enter into the BQI contract is required

   d) Scoring, ranking and enrollment will be for individual fields and/or pine forests >4.0 hectares in size

   e) Contracts will be issued per cooperator for all lands enrolled in a given focus area

   f) Annual habitat and compliance evaluations will be conducted by WRD biologists

   g) During each enrollment period WRD biologists will evaluate applications, contact cooperators, assist with developing plans, and score proposals

   h) Scored proposals will be forwarded to the BQI Headquarters where funding will be approved based on competitive ranking by focus area

   i) A minimum score is required for enrollment into the program , which will be set at the time of ranking

- j) Proposals which are not funded but which score > the minimum will be automatically rolled into the next funding cycle and will be ranked competitively with new applications
- k) Habitats established under a BQI Habitat Plan will be eligible for funding in future funding cycles

3) MONITORING PROGRAM:

- a) Cooperators enrolled in the monitoring program must agree not to conduct predator control, artificial feeding, and/or release of quail or other game birds in or around the contract area (unless otherwise recommended or approved by the WRD biologist)

4) INCENTIVES:

- a) No more than $15,000 will be allotted per property, cooperator, landowner and focus area per contract period (individual cooperator cannot exceed $15,000 during the 3 year contract period)
- b) Incentive contracts are for 3 years, and are renewable annually based on available funding and cooperator performance
- c) Incentive payments may be provided to cooperators for establishing and maintaining field borders, hedgerows, filter strips, center pivot corners, fallow patches, pine forest openings, linear pine forest practices, prescribed burning, thinning, herbiciding, and/or discing in pine forests.
- d) To be eligible for incentives individual crop fields and pine forest stands must be at least 4.0 hectares in size.
- e) To qualify for incentive payments all practices must be implemented and maintained according to BQI practice standards and recommendations as prescribed in the BQI Contract Management Plan
- f) Incentive payments are $148.26 per hectare per year for the establishment of dry land field borders, hedgerows/filter strips, fallow patches, and center pivot corners and $296.52 per hectare per year for irrigated field borders and hedgerows/filter strips
- g) Cost share payment is $6.07 per hectare per year for prescribed burning, disking, and/or herbiciding in pine stands with >30% sunlight
- h) Incentive payment is $148.26 per hectare per year for winter discing and/or herbiciding or $6.07 per hectare per year for prescribed burning in pine forest openings and linear pine forest practices
- i) Incentive payments are $6.07 per hectare per year for heavy thinning of pine forests (>40% sunlight on the ground at noon and/or <11.5m^2 per hectare BA). Prescribed burning will be required, with cost-share, to further enhance the quality of understory vegetation in the stand
- j) Cooperators can receive a maintenance payment of $98.84 per hectare per year for chemical or mechanical treatment of vegetation within enrolled areas, if recommended by a WRD biologist
- k) Funding for center pivot corners will be limited to 4.0 hectares per corner
- l) Funding for fallow patches will be limited to 4.0 hectares per field proposal
- m) When agricultural fields enrolled in the BQI program are in conservation tillage, cooperators will receive a Conservation Tillage Bonus Payment of $37.07 per hectare per year. Conservation tillage will be defined as any tillage system using a winter-grown cover crop and practicing residue management resulting in >30% ground residue throughout the summer. The payment will be calculated only for the crop field area in conservation tillage at the time of compliance and will not include hectares enrolled in other BQI practices.
- n) Funding for pine stand burning will not include hectares in other BQI practices

o) Funding will be annually distributed contingent upon successful implementation of habitat practices

p) Funds will be allocated based on the proposal score, which is a measure of habitat quality

q) Incentive payments are $98.84 per hectare for chemical treatment of invasive, exotic grasses when necessary to establish eligibility and will not be paid until successful completion of the first compliance evaluation

r) After WRD approval, funds will be allocated by the State Soil and Water Conservation Commission

5) HABITAT PRACTICES AND POINT VALUES

a) Field Borders - maximum of 150 points (25 points each)

   i) Must be at least 9.1 meters wide at narrowest point
   ii) Field borders may be stacked to double width (18.2 meters)
   iii) The number of field borders for a given field will be determined by the percentage of the total field circumference that the border occupies; one field border must equal 25% of the field with no segment <212.1 meters
   iv) Field perimeter covered by BQI practices (i.e., center pivot corners, fallow patches) will be subtracted from the field perimeter for the purpose of determining the percentage of field covered by field borders
   v) May require light discing during November - February
   vi) Cannot be used for turn rows, travel avenues, or hay or equipment storage
   vii) After establishment cannot be mowed, disced, burned, or treated with herbicides during the contract period unless recommended by the WRD biologist
   viii) Field borders can be established on the interior edge of fallow patches or center pivot corners.
   ix) Field borders may be established along the sides of existing forested hedgerows or filter strips
   x) Incentive payment is limited to the hectares included in the 9.1 meters field border or 18.2 meters stacked field border
   xi) To attain water quality points field borders must be within 9.1 meters of a watercourse

b) Hedgerows -maximum of 40 points (10 points each)

   i) Hedgerows must extend across entire length of crop field or connect to another acceptable habitat area, except for equipment travel avenues as approved by the WRD biologist and these are not to exceed 9.1 meters in width
   ii) Hedgerows must be at least 90.9 meters long
   iii) Hedgerows must be at least 181.8 meters apart
   iv) Hedgerows must be at least 9.1 meters wide at narrowest point along entire length
   v) Hedgerows may be stacked to double width (18.2 meters)
   vi) Hedgerows must be >30.3 meters from parallel field borders and/or filter strips
   vii) Hedgerows can be perpendicular or parallel to field borders and/or filter strips
   viii) Hedgerows must be established by light discing, fire or herbicides as recommended by the WRD biologist
   ix) Tree canopy cannot shade out more than 10% of the hedgerow area when determined at noon during the growing season

  x) Forested hedgerows (where tree canopy covers more than 10% of ground cover) may be renovated by reducing the tree canopy so that less than 10% of the ground cover is shaded out
  xi) Hedgerows cannot be used as turn rows, travel avenues or hay or equipment storage
  xii) Incentive payment is limited to area included in the 9.1 meters hedgerow or 18.2 meters stacked hedgerow

c) Filter Strips - maximum of 40 Points (10 points each)

  i) Filter strips must be at least 90.9 meters long
  ii) Filter strips must be at least 9.1 meters wide at narrowest point along entire length
  iii) Filter strips may be stacked to double width (18.2 meters)
  iv) Filter strips may include ditch banks but must be naturally vegetated with grasses, forbs, and shrubs for at least 9.1 meters on each side
  v) Filter strips must be >30.3 meters from parallel field borders and/or hedgerows
  vi) Filter strips can be perpendicular or parallel to field borders and/or hedgerows
  vii) Filter strips must be established by light discing, fire, or herbicides as recommended by the WRD biologist
  viii) Tree canopy cannot shade out more than 10% of the filter strip area when determined at noon during the growing season
  ix) Filter strips (where tree canopy covers more than 10% of ground cover) may be renovated by reducing the tree canopy so that less than 10% of the ground cover is shaded out
  x) Filter strips cannot be used as turn rows, travel avenues, or hay or equipment storage
  xi) Incentive payment is limited to area included in the 9.1 meters filter strip or 18.2 meters stacked filter strip

d) Center Pivot Corners - maximum of 40 points (10 points each)

  i) Must be at least one acre in size
  ii) Only eligible when connected by field borders or hedgerows
  iii) At least 70% of ground must be in direct sunlight at noon during the growing season to be eligible for enrollment ($148.26/ha/yr); however, corners established in thinned pines with >30% but <70% of the ground in direct sunlight may be managed using prescribed burning ($6.07/ha/yr)
  iv) To be eligible for incentives, corners planted to pines must be planted at <1,236 trees per hectare
  v) Wildlife plantings approved by the WRD biologist will be allowed on <25% of the site
  vi) Herbaceous ground vegetation must be established by light discing, fire, or herbicides as prescribed by the WRD biologist
  vii) Cannot be used as turn rows, travel avenues, or hay or equipment storage
  viii) Funding for corners will be limited to 0.4 hectares per corner
  ix) At the recommendation of the WRD biologist, discing or other prescribed treatments may be conducted on the corner hectares in the first contract year provided that the entire acreage is treated by the end of the second contract year. Incentive payments will be made for the entire corner hectares during each year of the contract period; however, if the cooperator fails to treat any portion of the site as directed by the BQI Contract Management Plan the contract will be voided and incentive payments will not be allocated.

e) Fallow Patches - maximum of 20 Points (5.0 points per hectare)

  i) Fallow patches must be part of, or in association with, an enrolled BQI field as determined by a WRD biologist; patches do not include center pivot corners
  ii) Fallow patches that are part of an enrolled field do not require a cropping history for enrollment; fallow patches that are in association with an enrolled field do require a cropping history
  iii) Fallow patches are permitted within the interior of LLCPA fields; fallow patches associated with a LLCPA field are not allowed
  iv) Fallow patches must be 0.4 - 4.0 hectares in size; total acreage in fallow patches cannot exceed 10 acres per proposal
  v) Fallow patches must be a minimum of 15.2 meters in width along their entire length
  vi) Tree canopy cannot comprise more than 30% of the patch. At least 70% of the ground must be in direct sunlight at noon.
  vii) Patches must be maintained by winter discing or as prescribed by the WRD biologist
  viii) At the recommendation of the WRD biologist, discing or other prescribed treatments may be conducted on the patch acreage in the first contract year provided that the entire area is treated by the end of the second contract year. Incentive payments will be made for the entire patch hectares during each year of the contract period; however, if the cooperator fails to treat any portion of a patch as directed by the BQI Contract Management Plan the contract will be voided and incentive payments will not be allocated
  ix) Plantings approved by the WRD biologist will be allowed on <25% in patches
  x) Fallow Patches must be >181.8 meters apart
  xi) Fallow Patches cannot be used for turn rows, travel avenues, or hay or equipment storage
  xii) After establishment, patches cannot be mowed, disced, burned, or treated with herbicide during the contract period unless recommended by the WRD biologist

f) Pine Forest Management - Understory management - maximum of 100 points (12.4 points per hectare)

  i) Predominately upland pine forests that do not include longleaf CPA
  ii) Must be at least 4.0 hectares in size
  iii) At least 30% (BA $<13.8\text{m}^2$ per hectare) of ground must be in sunlight at noon during the growing season to be eligible for enrollment
  iv) Pine forests that require thinning must be thinned before May 15th of the first contract year
  v) Pine straw raking will not be allowed at anytime or in any portion of the pine stand enrolled in BQI during the BQI contract period.
  vi) Burning, herbiciding, and/or discing must be conducted as recommended by the WRD biologist
  vii) At the recommendation of the WRD biologist, pine forest management may be conducted on the pine forest hectares in the first contract year provided that the entire contract area has been managed by the end of the second contract year. Incentive payments will be made for the entire contract hectares during each year of the contract period; however, if the cooperator fails to manage any portion of the pine forest as directed by the BQI management plan the contract will be voided and incentive payments will not be allocated
  viii) Hectares enrolled in openings and linear practices must be subtracted from total pine stand hectares to calculate points and incentives

g) Pine Forest Thinning - maximum of 50 points (2.5 points per hectare)

  i) At least 40% (BA $<11.5\text{m}^2$ per hectare) of ground must be in sunlight at noon during the growing season to be eligible for the heavy thinning incentive payment

- ii) Thinning must be completed before May 15th of the first contract year
- iii) Burning, herbiciding, and/or discing must be conducted as recommended by the WRD Biologist

h) Pine Forest Openings - maximum of 50 points (5.0 points per hectare)

- i) At least 30% (BA $<13.8\text{m}^2$ per hectare) of the ground within the pine forest must be in sunlight at noon during the growing season to be eligible for openings
- ii) Pine forest openings must be 0.4 - 2 hectares in size
- iii) Pine forest openings must be a minimum of 60.6 meters in width along their entire length
- iv) Tree canopy cannot comprise more than 10% of the opening interior
- v) At the recommendation of the WRD biologist, discing or other prescribed treatments may be conducted on the opening hectares in the first contract year provided that the entire hectares are treated by the end of the second contract year. Incentive payments will be made for the entire area during each year of the contract period; however, if the cooperator fails to treat any portion of an opening as directed by the BQI Contract Management Plan the contract will be voided and incentive payments will not be allocated
- vi) Plantings approved by the WRD biologist will be allowed on <25% of each opening
- vii) Openings must be >181.8 meters apart, must be included within the pine forest, and cannot comprise more than 40% of the pine forest stand

i) Linear Pine Forest Practices - maximum of 20 points (5.0 points per hectare)

- i) At least 30% (BA $<13.8\text{m}^2$ per hectare) of the ground within the pine forest must be in sunlight at noon during the growing season to be eligible for linear practices
- ii) Linear practices include privately owned roadsides, firebreaks, and borders surrounding pine forests
- iii) Must be at least 12.1 meters wide at narrowest point (measured from bole to bole)
- v) Parallel linear practices within pine forest stands must be >181.8 meters apart.
- v) Incentive payments are limited to area included in the 12.1 meters width
- vi) Linear practices must be maintained by winter discing or as prescribed by the WRD biologist
- vii) At the recommendation of the WRD biologist, discing or other prescribed treatments may be conducted on the linear area in the first contract year provided that the entire area is treated by the end of the second contract year. Incentive payments will be made for the entire area during each year of the contract period; however, if the cooperator fails to treat any portion of a linear practice as directed by the BQI Contract Management Plan the contract will be voided and incentive payments will not be allocated
- viii) Cannot be used for turn rows, travel avenues, or hay or equipment storage
- ix) After establishment, cannot be mowed, disced, burned, or herbicided during the contract period unless recommended by the WRD biologist
- x) Plantings approved by the WRD biologist will be allowed on >25% of the linear pine forest practices

j) Habitat Connectivity- maximum of 20 points (5 points each)

- i) Contract sites adjacent to areas (>20.2 hectares) of quality early successional habitat (for example thinned and burned woods and young pine forests with canopy spacing that allows >30% of the ground to be in sunlight at noon during the growing season)
- ii) Field borders entirely along all sides of crop fields and Longleaf CPA fields

- iii) Linear pine forest practices entirely along all sides of pine stands
- iv) Field borders established between crop fields and thinned and burned pine forests >4.0 hectares in size
- v) Field borders, hedgerows, or filter strips connecting two or more thinned and burned pine forests
- vi) Field borders connected by hedgerows in cropland fields and Longleaf CPA fields
- vii) Fallow patches connected by field borders, hedgerows, or filter strips
- viii) Enrolled pine forests adjacent to row crop agriculture or a LLCPA field enrolled in BQI
- ix) Using linear forest practices to connect to other BQI habitats

k) Habitat Bonus Points - maximum of 20 points for crop fields and longleaf CPA stands; maximum of 40 Points for pine stands

- i) An enrolled field is in conservation tillage (as defined in Funding section) during at least 2 of the BQI contract years - 10 points
- ii) When field borders, fallow patches, or filter strips are immediately adjacent to wetlands, ponds or streams - 10 points
- iii) When field borders are on more than one side of a thinned and burned pine forest (>0.4 hectares in size) that joins a crop field - 10 points
- iv) Fallowing of whole fields >16.2 hectares in the contract area (includes LLCPA fields) - 15 points Maximum of 5 points for planting approved foods per pine forest stand or field (>25% of enrolled BQI practices)
- v) Maintaining >50% sunlight on the ground (equivalent to >9.2m^2 per hectare BA) within a pine forest (excluding fallow practices) that is being proposed for enrollment - 10 points
- vi) Proposed pine forests that are >50% longleaf pine in tree species composition (excluding LLCPA fields) - 10 points
- vii) When cooperators have 2 or more agricultural fields, and/or LLCPA fields, and/or pine forests proposed to be enrolled in BQI - 5 points
- viii) Bonus for percentage of pine forest stand in managed fallow openings and/or linear habitat practices (20% = 10 points, 30% = 20 points, and 40% = 30 points)

l) Funding Bonus Points - maximum of 20 points

- i) 15% reduction in incentive payment - 5 points
- ii) 25% reduction in incentive payment - 10 points
- iii) 40% reduction in incentive payment - 15 points
- iv) 50% reduction in incentive payment - 20 points
- v) The payment reduction shall be applied to the Total Annual Incentive Payment (including incentive payments for all BQI habitat practices and the Conservation Tillage Bonus Payment); the reduction shall not be applied to payment for the chemical treatment of invasive, exotic grasses required to establish eligibility

m) Re-enrollment Bonus Points - maximum of 20 points

- i) Following 3 consecutive years of enrollment, fields may receive a re-enrollment bonus - 20 points.
- ii) When a cooperator who has been successfully enrolled for 3 or more years but is currently not enrolled and has maintained their BQI habitat at a level >55 points - 20 points

n) Additional Scoring and Contract Information

- i) Pine Forests should be separated into 2 or more stands whenever:
    - 1) there is age class or management change that results in significant difference in the way the stand will have to be treated for management and scoring; or
    - 2) the stand is completely split by a state highway or another cover type i.e. pasture, hardwood stand, river bottom etc. that averages >0.25 mile in width; or
    - 3) based on the WRD biologist it is in the best interest of the program or the cooperator to divide the stand,
- ii) When multiple cooperators occur under the same contract, then all cooperators must sign the "'Ownership and Payment Statement" and the "Signature Page for Contract with BQI Cooperator." Payments must be made per cooperator and the checks mailed accordingly. The payment due to each cooperator must be specified on the "Payment Due Cooperator(s)" page and cooperators must initial beside their respective payments
- iii) When multiple ownerships occur under the same contract, then all owners must sign the "Ownership and Payment Statement" giving the cooperator(s) permission to participate in BQI.

o) Determination of Previously Established Habitat

- i) Pine forests have been burned during at least 3 out of the last 5 years.
- ii) Field borders, and/or hedgerows, and/or fallow patches and/or center pivot corners are present and currently equal or exceed BQI standards.
- iii) Based on WRD biologist's opinion, the existing cover conditions provide quality habitat for quail and meet or exceed the current BQI minimum standards.
- iv) If at least two of these three criteria are met, it is likely that the property currently is under intensive quail management (pre-existing habitat) and therefore is not eligible for incentive payments through the BQI except where the cooperator agrees to include additional BQI habitat practices.

Appendix B. Georgia BQI Cooperator telephone survey conducted in 2004.

BQI Customer Service

Survey Question	Total N^a	Excellent (%)	Good (%)	Satisfactory (%)
BQI Cooperator experience rating	102	58 (57)	38 (37)	6 (6)
Quality service rating	102	77 (75)	24 (24)	1 (1)

BQI Cooperator Characteristics

Survey Questions	Total N^a	Yes (%)	No (%)	Not Sure (%)
Landowner	102	83 (81)	19 (19)	N/A
Primary farmer	102	65 (64)	37 (36)	N/A
Primary equipment operator	102	56 (55)	46 (45)	N/A
Have you ever quail hunted	102	92 (90)	10 (10)	N/A
Do you own bird dogs used for quail hunting	102	34 (33)	68 (67)	N/A
Do you plan to quail hunt this year	102	65 (64)	37 (36)	N/A
Have you hunted quail on property enrolled	102	52 (51)	50 (49)	N/A
Do you control hunting access	102	94 (92)	8 (8)	N/A
Do you allow quail hunting on land enrolled	94	33 (35)	61 (65)	N/A
Do you charge a fee for hunting privileges	94	25 (27)	69 (73)	N/A
Charge a fee for hunting privileges				
All game species	25	1 (4)	24 (96)	N/A
Deer	25	21 (84)	4 (16)	N/A
Turkeys	25	8 (32)	17 (68)	N/A
Waterfowl	25	1 (4)	24 (96)	N/A
Doves	25	5 (20)	20 (80)	N/A
Quail	25	2 (8)	23 (92)	N/A
Squirrels	25	1 (4)	24 (96)	N/A
Rabbits	25	1 (4)	24 (96)	N/A
Habitat practices pre-BQI				
Timber thinning	102	36 (35)	66 (65)	N/A
Prescribed burning	102	47 (46)	55 (54)	N/A
Weedy field borders	102	22 (21)	80 (79)	N/A
Weedy hedgerows	102	23 (23)	79 (77)	N/A
Weedy field corners	102	22 (21)	80 (79)	N/A
Winter discing	102	24 (24)	78 (76)	N/A
Planting food plots for quail	102	49 (48)	53 (53)	N/A
No practices were implemented for quail	102	21 (21)	81 (79)	N/A
Habitat practices post-BQI				
Timber thinning	102	40 (39)	62 (61)	N/A
Prescribed burning	102	56 (55)	46 (45)	N/A
Weedy field borders	102	96 (94)	6 (6)	N/A
Weedy hedgerows	102	82 (80)	20 (20)	N/A
Weedy field corners	102	81 (79)	21 (21)	N/A
Winter discing	102	75 (74)	26 (26)	N/A
Planting food plots for quail	102	63 (62)	39 (38)	N/A
No practices were implemented for quail	102	0 (0)	102 (100)	N/A

Appendix B cont'd. Georgia BQI Cooperator telephone survey conducted in 2004.

BQI Cooperator Characteristics

Survey Questions	**Total N^a**	**Yes (%)**	**No (%)**	**Not Sure (%)**
Cooperator perception of BQI impacts				
Improved farm environmental condition	102	93 (91)	1 (1)	8 (8)
Increased quail populations	102	83 (81)	1 (1)	18 (18)
Increased songbirds and other wildlife	102	84 (82)	1 (1)	17 (17)
Reduced soil erosion on land	102	70 (69)	20 (19)	12 (12)
Improved quail hunting	102	80 (79)	4 (4)	17 (17)

Additional Questions

Would implement BQI habitat practices without economic incentives (N = 102)

Yes: 24% No: 38% Some: 38%

Factors most influencing decision to participate in BQI (N = 102)

	N^a
Economic incentive payments	34 (33%)
Desire to improve quail populations	68 (67%)

Charge hectare/season to lease hunting privileges for wild quail on land enrolled in BQI (N = 87)

Less than $1.00	12 (14%)
$1.00 - $3.00	16 (18%)
More than $3.00 but less than $5.00	14 (16%)
More than $5.00 but less than $8.00	10 (12%)
$8.00 or more	35 (40%)

BQI practice affects on property aesthetic appearance (N = 102)

Improved appearance	64 (63%)
Detracted appearance	38 (37%)

[a] Number of people responding to the question(s).

Managing Quail in Arizona: Meeting New Challenges with Old Techniques

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I present an overview of past quail management in Arizona and the current direction of Arizona's Quail Program. Since the inception of Arizona's quail management program, management activities progressed from an era of intensive population and habitat data collection and habitat improvements, to one of more passive management. I explore the reasons for the de-emphasis of field activities and active management, and will discuss the changing face of quail habitat and quail hunters in Arizona. I will also discuss quail management issues related to hunter recruitment and retention, and the current social climate that makes annual population data collection and more active habitat management activities both desirable and necessary.

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Key words: active management, Arizona, *Colinus virginianus*, data collection, habitat loss, hunter recruitment, hunter retention, passive management, quail, research, urbanization

Introduction

Quail research and management in Arizona progressed from an era of active, field oriented activities (Brown 1989), to more passive management and research initiated only in response to controversy. Since the mid 1970s, hunter numbers have declined in Arizona as a percentage of the population, which is consistent with national trends (United States Department of Interior and Fish & Wildlife Service and United States Department of Commerce, Bureau of Census 2001). Arizona quail hunter numbers have also been trending downward, due to loss of access to hunting land, drought, and in some cases lack of information (Arizona Game and Fish Department 2006). The satisfaction of avid quail hunters with the Arizona Game and Fish Department's (the Department) quail management program also declined in response to more passive management (C.J. Biller, J. Levy, D. Lukens, T. Pfister, M. Rabe, personal communications). Their decreased willingness to work with the Department helped set the stage for what has been coined the "Arizona Quail Wars" (ca. 1995-2002).

Like many portions of the United States, Arizona is increasingly urbanized, with a human population growth rate twice the national average (United States Department of Commerce 2000). Sprawl surrounding urban centers results in fewer local places to hunt, and also contributes to problems of hunter retention (Schulz et al. 2003). More importantly declines in small game hunting participation have exceeded those for big game (United States Department of Interior and Fish & Wildlife Service and United States Department of Commerce, Bureau of Census 2001). The lack of local places to hunt has impacted urban hunter participation, and those available habitats adjacent to urban centers cannot meet the needs of all urban hunters (Schulz et al. 2003). This results in declining participation and license sales (Schulz et al. 2003), then ultimately reducing funding for conservation programs. Since local small game hunting represented the traditional recruitment mechanism for most youth hunters in the past (Adams et al. 2004), loss of local hunting areas not only impacts retention of existing hunters, but

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recruitment, as well.

Today's hunters require more information in order to be successful in a shorter period of time due to competing interests and a desire to maximize opportunity for success during their increasingly limited free time. While some level of success is important for hunters to continue hunting (Ortega y Gasset 1985), early and consistent hunter success may be more important to today's hunter, particularly those new to the activity (Duda et al. 2003). Survey results continue to show that being outdoors and sharing the experience with family or friends are key reasons they choose to hunt, as well (United States Department of Interior and Fish & Wildlife Service and United States Department of Commerce, Bureau of Census 2001, Arizona Game and Fish Department 2005).

Like many other state wildlife agencies and nongovernmental organizations (NGOs), the Arizona Game and Fish Department is increasing its efforts regarding recruitment and retention of hunters, both to maintain and increase funding for conservation and to preserve America's hunting heritage. Declining participation, reduced free time among potential hunters, increasingly sophisticated dedicated quail hunters, habitat degradation, and declining habitat connectivity have led to a time when a more active management program is again called for in Arizona. This paper documents past quail management efforts, and current efforts aimed at positively influencing quail conservation, management, and recruitment and retention of hunters.

Arizona quail management history 1940-1987

Habitat management

The Arizona Game and Fish Department was actively protecting and enhancing quail habitats throughout the state as early as 1939. Prior to this, quail (primarily Gambel's quail, *Callipepla gambeli*) were considered to be unimaginably abundant, and management efforts often involved removals of "nuisance" quail from agricultural areas (Griner 1940*a*). These animals were often used in translocations to other parts of the state throughout the 1930s and 1940s. Concerns did exist, however, and Gorsuch (1934) was warning about the effects of overgrazing and declining quail as early as the late 1920s and early 1930s. Initial habitat protection and enhancements by the Arizona Game and Fish Department began in Cochise County in 1939, including inventories of existing habitats, roadside surveys of quail populations, and annual production surveys (Griner 1940*e*). Areas identified as key habitats were fenced (4-45 ha, 10-110 acres), and data were collected annually in these "Quail Restoration Plots" from 1940-1945. Extensive inventories of quail habitats and quail populations throughout the state followed initial efforts in Cochise County, but restoration plots were installed only sporadically (Griner 1940*c*,*g*,*f*,*b*,*d*). The post-war era led to shifting priorities and increased workloads for Arizona Game and Fish Department personnel that resulted in abandonment of these restoration plots, although it was recommended that a portion of those be retained in Cochise County (Wright 1951). Additional quail habitat protection efforts resulted in the purchase of tax delinquent properties along the Gila River between Phoenix and Yuma (Brown 1989). These properties remain a portion of the Arizona Game and Fish Department's Wildlife Management Area system and remain important habitats for Gambel's quail.

Given the arid nature of much of Arizona, early managers believed the addition of free water would increase quail abundance, a question that lingers through today. Construction of artificial water sources for quail began in 1946 in central Arizona (Kimball 1946). Experimental research on water requirements and the influence of artificial waters on quail abundance began in 1957 and continued through 1963. Through experimental water closures, Smith and Gallizioli (1963*a*) determined the addition of free water did not increase hunter success or Gambel's quail abundance. This research suggested providing additional waters through guzzler development may actually be detrimental to quail populations since they artificially concentrate quail, making them more vulnerable to harvest and disease. Following these findings, "gallinaceous guzzler" construction was discontinued although the Department continues to actively provide artificial water catchments for big game. Active manipulation of habitat ceased for Gambel's quail following discontinuation of guzzlers, as it was determined the protection of large blocks of unaltered native habitats would have a more positive impact on populations.

Little habitat effort was directed at Arizona's other quail species from 1940-1987. The work of Griner (1940*e*) in Cochise County benefited scaled quail (*Callipepla squamata*) more than Gambel's quail, since retention and enhancement of desert grassland habitats was a primary goal of these restoration plots. Efforts began to evaluate habitat requirements of scaled quail of southeastern Arizona in 1967 and plans were made to translocate this species to grassland habitats in other parts of Arizona. However, following an analysis of habitat conditions, life history, and precipitation patterns, Brown (1970) recommended that this program be terminated.

Montezuma or Mearns' quail (*Cyrtonyx montezumae mearnsi*) were considered to be rare following the intense period of livestock grazing and drought from the 1880s to after World War II. Rangelands recovered somewhat during the post-war years and Mearns' quail populations responded favorably (Brown 1989). Efforts to gather more habitat and distribution data followed a request to open a season for this species in 1960 (Yeager 1967). No active habitat work was conducted for this species beyond making recommendations to the USDA Forest Service concerning livestock grazing management.

Quail population survey

A variety of techniques were tested and employed in Arizona to determine annual quail populations and to create an index of fall hunt success. Gambel's quail summer roadside and brood counts were conducted annually (2,200-3,200 km, 1,400-2,000 miles, annually) from 1941-1963 to assess statewide populations prior to the hunting seasons (Smith and Gallizioli 1963*b*). Scaled quail data were collected incidentally along these routes, but did not represent a serious effort for this species. Total number of quail observed on summer routes was slightly correlated with the percentage of juveniles in the harvest ($r^2 = 0.22$) and was a poor predictor of hunt success. Links between winter precipitation and juvenile production were suspected, but had not been thoroughly evaluated. Using data from 1941 - 1961, winter precipitation showed a high correlation with percentage of juvenile Gambel's quail in the harvest ($r^2 = 0.70$), but, again, was not a good predictor of overall hunt success, since hunter success also depended on population carryover and local cover conditions (Smith and Gallizioli 1963*b*). However, in the 1940s, hunting seasons were closed if the observed juvenile per adult ratio fell below 2.1:1 (Brown 1989).

Gullion (1954) in Nevada, and Hungerford (1960, 1964) in Arizona conducted analysis linking winter precipitation and growth of green forbs rich in Vitamin A. In 1960, the link between calling intensity, young produced, and winter rainfall was firmly established (Hungerford 1960), further bolstered by Hungerford (1964) that linked winter precipitation with green forb production and corresponding gonadal development and reproduction. Following an 7-year period of analysis (Kufeld 1964), spring Gambel's quail call count surveys replaced summer roadside surveys in 1964, and were conducted statewide until 1987 (L. Ordway personal communication). Research (Smith and Gallizioli 1965) suggested the number of calls recorded at 0.5 mi intervals along a 20 mi route proved a good predictor of nearby hunter success. However, these routes were discontinued throughout much of the state in response to increased field personnel workload, shifting priorities, and the relationship between precipitation and quail production, despite recommendations from earlier research that precipitation data alone was a poor predictor of hunter success (Smith and Gallizioli 1963*b*).

Outside of attempts to gather limited population data, fewer research and management actions were directed at the other hunted quail species in Arizona, scaled quail and Mearns' quail, since they represent a relatively low percentage of annual quail harvest in Arizona. D. E. Brown did the first serious population survey work with scaled quail in Arizona (Brown et al. 1978). Effects of precipitation on nesting success were poorly correlated, and nesting success for this species was found to be fairly consistent, suggesting variation in hunt success was more dependent on population carryover.

Mearns' quail population surveys began in response to the first open the season (2 days) in 1960. Roadside and pointing dog surveys were conducted sporadically to assess sex and age ratios, numbers, life history and general distribution. Serious work to develop a survey technique for Mearns' quail did not occur until the late 1990s (Bristow and Ockenfels 2000).

Quail harvest survey

Indices or estimates of population and determination of harvest levels for game species continues to remain an annual data need for state wildlife agencies, and are increasingly expensive and/or difficult to obtain (A. Munig, B.Wakeling, M. Rabe personal communication). Efforts to gather quail harvest data in Arizona began in 1940, using check stations in a few local, popular hunting areas (Eicher 1943).

Prior to standardized Arizona quail seasons in 1979 (Brown 1989), quail season length and bag limits were adjusted annually in response to quail abundance indices and political considerations.

Consistent collection of harvest data at check stations did not begin until 1951 (Gallizioli 1955), although these efforts were typically tied to a research project and were not conducted solely for management purposes. Check stations operated from 1951-1960 were used primarily to assess harvest impacts on quail populations, results of which suggested regulated harvest had little impact on quail abundance (Gallizioli and Webb 1961). Two check stations for Gambel's quail, both in the Department's Southeast Region (Region 5) have been conducted annually from 1980 to present (J. Heffelfinger, personal communication). Other harvest survey methods employed during this period included daily hunter report cards, and wing barrels. Wing barrels to collect Mearns' quail harvest data were initiated in the 1960s and continue through the present. The Department's mailed hunt questionnaire system was initiated in 1961 (Arizona Game and Fish Department unpublished data), began providing reliable estimates of hunter statistics and harvest by 1965 (Brown 1989), and continues to be in use today.

Arizona quail management history 1988 - 2002

Passive management

By 1988, Arizona's quail management program could best be described as passive, although a few annual active programs continued. Because of competing priorities, conflicting opinions on importance of field data, declining revenues for game management, and increasingly diversified workloads, statewide field data collection was discontinued. Seasons and bag limits were standardized in 1979 and hunters were informed that precipitation dictated quail abundance, seemingly negating the need for other sources of information. This began to set the stage for controversy. Standardized seasons and bag limits, an over-reliance on annual precipitation data as the main index for season forecasting, less contact with constituents in the field, and the appearance of doing little to protect or enhance quail habitats created a rift between the Department and our core support group, namely the "dedicated quail hunter." Indices of Gambel's quail population were discontinued entirely throughout much of the state, with the exception of 3-5 annual Gambel's quail routes run only in southeast Arizona, which is not representative of the state. Little or no data were collected for scaled quail. Controversy in the late 1990s and early 2000s surrounding harvest impacts to Mearns' quail instigated the creation of a coalition of hunters and guides that acted as a "watchdog" to Department's quail management actions. In an effort to reengage these constituents volunteers from a variety of NGOs and local Mearns' quail guide services were employed to collect pre- and post-season Mearns' quail data. However, these efforts were discontinued after a few years due to variability between observer and dog abilities.

The Department also did little in the way of quail research during this time period. Most research initiated as a result of the controversy, such as the research of Bristow and Ockenfels (2000, 2002, 2004), and Bristow et al. (2005).

Conflicts Created

During this era, several issues arose that created problems for the Department, alienated some constituents, and reduced program effectiveness. Quail hunter opinion of the Department's standing as a leader in quail conservation and management declined. A large portion of the quail hunting public believed the Department had lost touch with local quail abundance, impacts of harvest, and habitat condition. Worse still, the Department was looked at as doing little actual work to protect and enhance quail habitats (J. Levy personal communication). While these issues were being actively worked on by Department employees, information regarding efforts was not well coordinated and was slow to be disseminated to constituents. The controversy culminated in the Department hosting a Quail Symposium in 2002. Efforts were made to include as many constituents as possible, and quail management experts from around the country were invited to assist the Department in assessing and potentially making recommendations for improvement of quail management. The symposium pacified some constituents, further alienated others, and ultimately changed little regarding management activities.

Arizona quail management history 2003 - present

Current challenges

Arizona quail species and quail hunters are facing enormous and seemingly insurmountable challenges. Arizona has one of the fastest growing human populations in the nation (e.g. Phoenix is the 6th largest city in the US and its growth rate is twice the national average), and growth will likely continue at a high rate, with a current population of >5 million more than doubling by 2030 (United States Department of Commerce 2000). Illegal immigration (conservatively estimated currently at over 0.5 million) from Mexico and other Latin American countries is accelerating, and habitat impacts in rural southern AZ are increasing dramatically due to this and other cross-border activity. Residential development is rampant, seemingly with little or no planning, and quail habitats and habitat connectivity are being lost permanently throughout Arizona, particularly in core Gambel's quail range of central Arizona. Grazing management continues to represent a significant habitat concern throughout much of Arizona, but cannot match the threat of residential development. Mearns' quail habitats are relatively secure from development due to their federal land status, but isolation of "sky island" populations will increase as low elevation habitats between mountain ranges fill with houses. Scaled quail are also being impacted by increased development, as well as habitat conversion and range management practices. Maintaining wildlife habitats in Arizona for all species will require increased political clout and support for wildlife managers and management activities.

Hunter recruitment, hunter retention, and Arizona quail management

In Arizona, declines of upland game hunters is greater than that for big game (Arizona Game and Fish Department 2006). However, Arizona has a limited big game resource. Big game licenses are issued through a competitive draw, and applicants far outweigh available permits (Arizona Game and Fish Department 2006). To retain unsuccessful hunters, the Department is exploring ways to increase interest in upland game hunting since: 1) upland game opportunities are more consistent, 2) these species represent the traditional recruitment point for youth hunters, and 3) upland opportunities may serve as a bridge during years hunters are unsuccessful during the big game permit drawing (Arizona Game and Fish Department 2005).

Since Gambel's quail and mourning doves (*Zenaida macroura*) are the most widespread and abundant upland species in Arizona, the Department's Hunter and Shooting Sports Recruitment and Retention Team (HRRT) recommended that significant effort be placed on increasing the popularity of quail and dove hunting for both recruitment and retention efforts. Much of the emphasis will be placed on increasing the interest, knowledge, and abilities of potential and existing quail hunters, as well as their active involvement in conservation efforts.

Current direction

The Department's quail management program is in the process of returning to an era of more active field activities, including increased collection of population, harvest, and habitat condition data. These efforts place our personnel in the position of specific evaluations regarding quail species, versus the more casual approach of gathering these data incidental to other activities. Increased field presence during hunting seasons, opening weekend check stations, and survey efforts allow for increased contact with customers during critical time periods, and can improve relations with the customer even during years of lower quail abundance. Our current efforts have two ultimate objectives: 1) increase available funding for quail conservation, and 2) increase political clout by increasing constituent base. Both objectives are intimately associated with the Department's efforts regarding retention and recruitment of hunters.

The following assumptions apply to current and future activities: 1) increased customer service equals increased constituent loyalty, 2) increased constituent loyalty will translate into more political, financial, and involved support for conservation, and 3) increased field contact elevates Department's position as the state's quail management authority.

Our efforts are gradually increasing as funding allows. Increased efficiency is a key to increased data collection and information dissemination, and we are exploring increased use of citizen scientists to meet data needs. Increased use of citizen science can increase our support for conservation, and offers unique opportunities for increased conservation education, and an opportunity to foster public ownership of Arizona's quail management program.

The quail program continues to monitor and provide local precipitation information to hunters. We are now bolstering these data with local population data, including more call-count information, regional brood observations, and other relevant information. Efforts are underway to provide user-friendly local scale quail forecasts, improving our opportunities for successful hunts by our constituents. In 2003, we began increasing the number of check stations we operate during the opening weekend. These serve not only as a source of harvest and hunter demographics data, but also represent an excellent opportunity to increase constituent contact in field settings. Efforts to increase other data collection, including the increased use of wing envelopes, wing barrels, and information dissemination have all been met with positive feedback.

We are also increasing our efforts to enhance and protect quail habitats. Current activities involve working with municipal and county zoning boards in an attempt to ensure the needs of quail, other wildlife, and hunters are considered in community development. We also work closely with state and federal land management agencies in an effort to apply management prescriptions that benefit quail species. Active habitat manipulations are increasing, as funding sources can be located. As recommended by the HRRT, intensive manipulations (including farming practices) on Arizona Game and Fish Commission Wildlife Management Areas near urban centers designed to enhance dove and quail populations are increasing. These areas will serve a vital role in upcoming recruitment and retention efforts, including Department hosted small game hunting workshops and youth small game hunts. While competition for habitat funds in increasing, we are currently achieving success in desert grassland habitat enhancements for scaled quail at a landscape scale by pursuing partnership opportunities that leverage additional monies, including those destined for other grassland obligates.

Further application of more active management in Arizona will require a significant paradigm shift among agency leaders and wildlife administers regarding the need for field data and increased priority placed on habitat enhancements for upland game. Habitat funds in western states have traditionally been spent primarily for big game projects, particularly during the past thirty years. While population data are not a necessity for establishment of annual statewide seasons for quail species (Guthery et al. 2004), and precipitation ultimately determines quail abundance (Brown 1989, Engel-Wilson and Kuvlesky 2002), the data do have significant value to our hunting customers, and to the Department for relationship building and maintenance. Positive relationships with all quail hunters, and increasing the number of hunters will increase the conservation community's ability to positively influence the political system regarding wildlife and open space conservation.

Potential concerns and solutions

Increased hunter recruitment efforts in Arizona are being met with some resistance among Arizona's more ″avid″ or ″greedy″ hunters (Engel-Wilson and Kuvlesky 2002). Concerns exist regarding providing enough information to new or occasional hunters for success, and the potential for further alienation of more dedicated and actively engaged ones (e.g. by giving out ″favorite spots″). The Department will also be viewed as a glorified guide service if very specific hunting locations are provided. All our efforts should encourage field skills, educate, build conservation coalitions, and minimize divisiveness. Offering opportunities for involvement in data collection and program implementation to all Arizona quail hunters may provide the cohesiveness necessary to be ultimately successful in Arizona quail conservation. Increased customer loyalty will ultimately pay dividends in increased financial and political support for quail conservation programs. Rapidly increasing urbanization in AZ makes public support and corresponding political clout increasingly important to maintain open space, quail habitats and areas that can be hunted.

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Restocking

A Review of Gray Partridge Restocking in the UK and its Implications for the UK Biodiversity Action Plan

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The gray partridge *Perdix perdix* has been the subject of many re-introduction projects throughout the world. In earlier attempts many releases simply aimed to increase the number of individuals for harvesting. This is very different from a conservation project aiming to establish a self-sustaining population. In recent decades, the gray partridge has declined severely in abundance and it is a species of conservation concern throughout Western Europe. Until now, gray partridge releasing projects have mainly focused on releasing large numbers of captive-reared individuals, of which few survive because of heavy predation and low breeding success. We reviewed the scientific and gamekeepering literature, and found that nevertheless a number of traditional methods for rearing and releasing gray partridges exist. Although these have primarily been developed to supplement existing wild stocks to produce shootable resources, some can be re-used today for conservation purposes. The most promising system for producing birds for re-introduction and supplementation purposes is to obtain eggs from a reliable source, hatch and rear the chicks under bantams to eight weeks of age, then foster to failed pairs of wild gray partridges. A less labour-intensive alternative is to hatch and raise chicks under artificial heat and foster these to unsuccessful wild pairs. Obviously these two systems are dependent on the presence of local free-living wild birds. If no pairs of wild gray partridges are present it is necessary to establish a founder population first. We see two methods to achieve this goal, the release of coveys in autumn or of pairs in spring. An important pre-requisite to any restocking scheme is appropriate management including the provision of suitable habitat for feeding and nesting and the control of predators, otherwise restocking is unlikely to lead to long-term establishment.

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Introduction

Prior to the Second World War, the rearing and releasing of gray partridges was not commonly undertaken within the partridge's natural range. Good partridge management, which primarily concentrated on controlling predators, was sufficient to ensure that enough birds survived to provide a shootable resource. The farmed environment at the time was ideal for gray partridges, providing a range of naturally occurring nesting sites and good food availability. Consequently, gray partridges were one of the most common and widespread farmland birds (Potts 1986). In the UK, a number of methods for rearing and releasing were developed to ensure that this resource was even larger than it would be if left to nature, and imported ("Hungarian") eggs and birds were used solely to introduce new blood.

In the UK, agricultural intensification from the 1950s and a reduction in predation control resulted in huge declines in wild gray partridge numbers (Potts 1980, 1986) of 89% over the last 30 years accompanied by a 25% contraction in range during the same period (Gibbons et al. 1993, Baillie et al. 2005). As a result, the species has been recognised by the UK government as a Biodiversity Action Plan (BAP) species and as such has specific targets relating to increases in population size that must be met by 2010 (Anonymous 1995). During this decline phase, a number of estates tried rearing and releasing gray partridges on a large scale, whilst most oth-

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ers switched to the easier and more cost-effective releasing of pheasants and red-legged partridges and ceased active gray partridge management. Having identified the causes of the decline in the UK (Potts 1980, 1986), research sought to produce management options targeted at reducing the damaging effects of modern agriculture (Rands 1986, 1987, Sotherton et al. 1988). Many of these options are now part of government-funded agri-environmental schemes. Consequently, not only do we know what is needed to restore the fortunes of the gray partridge, but government has largely provided the means to do so.

A number of estates have adopted partridge-friendly agri-environmental options, but owing to the absence or low numbers of gray partridges occurring in the wild, the birds have not responded. How then to establish partridges in a sustainable fashion that will contribute to achieving the aims and targets of the species' action plan?

In this review of traditional and current rearing and releasing systems, which could potentially be used to contribute towards reaching the BAP targets, we seek to identify which method or combination of methods would be most suitable to answer this question.

The review was conducted by searching through old game management books, the scientific literature, and through formal and informal interviews and discussions with gamekeepers (see the References and Acknowledgements sections for the publications and people consulted).

Traditional rearing and releasing methods

Traditionally most gamekeepers left their gray partridges to take care of themselves. Only estates that used farming methods that resulted in large nest losses, for example hay and silage cutting or heavy grazing by stock, intervened and interfered with the natural breeding of the partridges. Where this happened, most techniques for rearing and releasing were variations of one of the four systems outlined below.

The Euston System

This system was developed to overcome the problem of hen and nest losses during the incubation period (through predation, disturbance, trampling, etc.), to spread the date of hatching and to increase genetic heterogeneity. As many gray partridge nests as possible were found and mapped. Nests that were regarded as truly safe were left to hatch naturally. For other nests, once they contained 5 eggs, the eggs were removed and replaced with matching wooden eggs. Further natural eggs were then removed on every other day until all natural eggs were removed and replaced with wooden ones. When the female gray partridge started incubating the wooden eggs, the date that incubation started was recorded. For nests that were in unsuitable positions all the eggs were removed and the nest destroyed, in the hope that the female would nest again elsewhere. The removed natural eggs were placed under broody bantams and monitored daily. Once the eggs started chipping, they were carefully replaced under the incubating female gray partridge, after first removing the wooden eggs. Generally it was possible to swap the eggs without disturbing the female, but if she did leave the nest, she was likely to return very quickly. It was believed that chipping eggs could be returned to an incubating female between 14 to 21 days after she started incubation, and the time of returning eggs should coincide with a period of good weather. Some gamekeepers would exchange eggs, prior to starting their incubation, with neighbouring estates to introduce "new blood" to the estate.

This system was successful in maintaining and enhancing gray partridge numbers to produce a shootable resource, but it was not intended to be used for restocking. It has not been used on a large scale in the UK for many decades (at least 60 years) and documented evidence of its success is not available in the modern literature, although a number of historical books state that it was highly successful (e.g. Alington 1910, Wormald 1912, Portal and Collinge 1933). It was very labour-intensive, however, and required a high degree of knowledge about the nesting behaviour of partridges. To really provide benefit for the shooting estate there needed to be a large resident stock, otherwise the effort involved was not justified. Additionally, this system increased only the number of nests that successfully hatched eggs and provided no additional benefits to the chicks. The key to success for this system was the widespread availability of suitable chick-rearing habitats, with an abundance of good invertebrate food. The system offered no protection against inclement weather, which can have a major impact on chick survival. For these reasons, the use of the Euston system is not really viable today.

The French or Continental System

This system was developed to allow gray partridges to rear their own young within a controlled system that protected them from predation, poor food availability during the first few days after hatching, and to some extent, the weather. Fundamental to the French system was the construction of a suitable partridge enclosure, which consisted of a large central area, maybe as large as 40 m by 60 m, with a number of smaller pens arranged around it. The central enclosure and the side pens contained bushes, fir branches, tussocky grass and a good supply of grit, water and food. Electric fencing and numerous traps was placed around the enclosure to prevent predation. Up to 60 true pairs (i.e. equal numbers of males and females) were introduced into the central enclosure, usually in November. Traditionally these were "Hungarian" birds, these being hand-reared birds usually imported from Hungary, but not exclusively. When introduced into the central enclosure each bird would have had one wing tied, or "brailed", to prevent the bird from flying and damaging itself. A gamekeeper would watch the birds and as soon as pairing was observed he would quietly drive the pair into one of the side pens where it would be left to produce eggs and hatch young. Pairs were never forced as it was felt that an "unnatural marriage" would never work. Any unpaired birds were released from the central enclosure into the wild, or retained as stock birds. Each female was permitted to lay up to 25 eggs. Any surplus was removed and incubated under broody bantams; the resulting chicks were either released when fully grown, or reintroduced to the penned birds to make up a brood. The penned pair meanwhile was left to incubate its clutch, kept within the pens until the chicks were 3-4 days old, then the adults and brood were released onto the estate. During bad weather, the adults and their chicks would be held in captivity until it improved.

Compared with the Euston System, this system provided assistance to gray partridge chicks during the first few days after hatching, when they are most susceptible to poor weather and limited food supply. Additionally it provided an opportunity to rapidly increase stock over a relatively short period, and as such provided an opportunity to restock as well as provide a shootable resource. Again it is difficult to find documented evidence of this system's success, but one gamekeeper reports that to produce a large stock this system is the best (Portal and Collinge 1933). Like the Euston System, this system was also very labour-intensive and required a dedicated gamekeeper to produce the pairs and a lot of knowledge to induce them to breed successfully.

The Montebello System

The French System required the construction of a large purpose-made enclosure and a gamekeeper to watch for paired birds. The Montebello System overcame the need for a specially constructed enclosure and the commitment of a gamekeeper's time, by forcing the pairing of a male and female. At the start of the rearing process, a forced pair was put into a small breeding enclosure, with the hope that they would accept each other and settle down to breed. Traditionally, the system would use either wild-caught pairs or a combination of a wild-caught bird and one from reared stock. The pen contained all of the requirements for partridge breeding, including food, water, grit, vegetative cover and a suitable nesting area. The pair of partridges was left to undertake egg laying and incubation. Once the chicks were 3-4 days old, they were released with the adults into suitable habitat close by.

This system was very similar to the Continental System, but overcame the problems of requiring a large, specially constructed pen and a dedicated gamekeeper. It still required the holding of birds in pens, with its associated problems of preventing predation, and was dependent on the ability of captive-held partridges to produce eggs, undertake incubation and rear young. This ability to act as suitable parents whilst in captivity is something that is dubious today (see later).

Traditional hand-rearing of partridges

Historically gray partridges were not reared and released in the sense that we would regard the rearing of pheasants and red-legged partridges today. Generally, a gamekeeper would acquire a few eggs from badly sited or predated nests, from friends, as a result of surpluses from the Euston or French System or as bought-in "Hungarian eggs". Foremost in the gamekeeper's mind was cleanliness and disease prevention. All equipment associated with rearing was scrupulously cleaned, including broody boxes, coops, feeders and drinkers prior to the start of the breeding season. Broody hens and bantams were all treated for scaly leg and were dusted with insect powder. Each hen or bantam was given up to 30 partridge eggs to hatch, with some being removed for the Euston System or some other purpose, ultimately leaving the hen with 18 eggs. Once the eggs were hatched and whilst the chicks were between 12-24 hours old, the hen and 16 chicks were moved to a rearing field and placed in a coop (i.e. a box that contained the hen, but allowed the chicks to move freely in and out of the box via wooden slats or bars). Excess chicks were divided between broods to make numbers up to 16. Coops were positioned about 80 m apart and had fir branches positioned near them to provide cover. The coops were kept in an open field, which required rigorous predator control especially at night, even though the coops were shut each night. Each coop was moved onto fresh ground daily, when the hen was removed from the coop and allowed to eat and drink whilst being tethered by one leg. The chicks were fed three times a day by scattering an appropriately prepared mixture of egg yolk, scalded biscuit meal and barley flour outside the coop. Ants and their eggs were also offered to the chicks once a day. Once the chicks were 4 days old, chopped onion and boiled rabbit was also added to the feed. Until the age of 8 weeks various other ingredients were added to the food, including maize, rice and egg white. When the chicks were 8 weeks old, the coops were moved to a field edge and arranged in pairs, at about a half mile distance from the next nearest pair of coops. Feeding was reduced to three times a day, coops were still moved daily, but were not shut up at night. At about 9 weeks, one of the hens was removed from the pair of coops and a week later the other hen was removed. The coops were left in place, but were finally removed by mid-September. Usually a barren wild pair would come and adopt the chicks or the chicks would disperse naturally.

This incredibly labour-intensive system would have undoubtedly produced a shootable resource by the start of the shooting season, and obviously the more birds reared the larger the resource. However, what is not known, or recorded, is how successful this system was in supplementing the existing wild stock or in restocking areas where only a small residual wild population remained. It is likely that some of the released birds would have survived over winter and bred the following year, principally because they would have mixed with true wild birds, but how the reared birds would have survived in the absence of a wild source is unknown.

Current rearing and releasing methods

Over the last 10-20 years, very few estates in the UK have used traditional methods to rear and release gray partridges for restocking. However, in East Anglia and a few isolated pockets elsewhere in the country, some estates have developed techniques that are essentially modifications of the traditional methods, but which use improved technology and a better understanding of gray partridge behaviour.

Modified Montebello System

Over the last 10 years a number of estates in Norfolk (Eastern England) have been using a modified version of the Montebello System. A pair of gray partridges is kept in a small enclosure and allowed to breed, but chicks are retained up to the age of at least six weeks before release (K.A. Blake, unpublished). This sought to overcome the problems of early chick losses associated primarily with limited food availability in modern-day arable habitats. In order to simplify management of the birds, some estates tried keeping the paired pens in close proximity on a rearing field before moving the adults and chicks to suitable areas for release. Pairs apparently spent a lot of time calling to one another and pacing up and down the sides of the pens in an attempt to get together. This seemed to confuse the pairs, which did not settle and in some cases started fighting. To overcome this, pens were then sited in natural habitats away from the other pens so that birds would not call to each other. Although this overcame the problem, occasionally wild single male birds would be attracted to the pens, which resulted in fighting and poor fertility. Pairs were formed by using either reared birds or a mix of wild and reared birds. The problems of fighting within the pairings, poor fertility, and the use of reared birds often meant that birds did not incubate the eggs correctly and produced only small broods of young. To overcome this, some estates exploited the natural instinct of adult gray partridges to adopt young partridges. The young were produced by hatching the eggs produced by the penned birds under bantams or by purchasing day-old chicks from game farms. The young were often adopted immediately after being introduced to the penned partridges. Another modification to the traditional Montebello System arose from the desire to retain the adult partridges as stock birds, for use in future years. To do this the adult partridges were taken away from the chicks at approximately six to eight weeks. The chicks were then moved to small pens sited within the territories of unsuccessful wild pairs, where they were released in the hope that they would be adopted.

Although estates that used the modified Montebello System reported initial successes, the problems outlined above led to the majority of estates moving to the system described in the next section. Brood production of the pairs held under this system varied from about 60% to 100% depending on the origin of the parents (K.A. Blake, unpublished). Pairs that were made up of a wild male and a captive-reared female had the highest rate of brood production, those that were from purely reared stock had the lowest brood production, and the pairs that were made up of purely wild birds were mid-way between. Mean brood size followed a similar pattern and ranged from 2.8 ± 1.0 (SE; $n = 7$) young per pair for reared pairs to 11.0 ± 1.4 (SE, $n = 12$) for mixed pairs.

Fostering to wild barren pairs

One of the most commonly undertaken methods of rearing and releasing young today has been fostering to wild pairs. For this system, eggs or young are reared to the age of about six to eight weeks, taken to the territory of a failed wild pair and released in the expectation that they will be adopted. Sources of eggs include those picked up from the wild, produced by stock birds or occasionally obtained from game farms. These eggs are either hatched and reared by bantams or hatched in an incubator and reared under an electric brooder. Once these chicks are at least six to eight weeks old they are formed into broods of 10-15, placed into small pens and moved into the territory of an unsuccessful wild pair. Some gamekeepers place these pens within a standing crop, whilst others place them in the open. Once the wild pair is seen close to the pens, the young are released and taken away by the wild pair. Occasionally a few of the young are released from the pen, so that those outside call to those inside the pen, thereby potentially increasing the attraction of the young birds to the adults. Usually the released young remain in coveys with the adults, but occasionally coveys have been known to merge.

The survival of these birds after release and their breeding success in subsequent years appears to be very good. This method has been scientifically evaluated in Switzerland and compared to other releasing techniques mentioned above, in the course of a re-introduction scheme. The Swiss scientists compared the monthly survival rates of individuals originating from three different release strategies: 1) translocation of wild adults, 2) releases of captive parent-reared adults and 3) captive parent-reared chicks that were fostered to wild pairs. Locally born offspring (second generation birds) from the established breeding pairs served as a control group. The fostered birds had the highest monthly survival rate (0.86 ± 0.03), followed by the translocated birds (0.82 ± 0.06) and by the adults reared in captivity (0.70 ± 0.06). The high survival rate of fostered birds was most likely due to acquiring knowledge about predator avoidance from their experienced parents (Buner and Schaub 2008). However, because that study was based at a single location with relatively low sample sizes, further research involving larger sample sizes and multiple study sites would be appropriate.

Modern-day rearing and releasing

The two previously discussed modern-day techniques have been primarily aimed at increasing stock levels and not necessarily enhancing the shootable resource. All of these techniques are labour-intensive, requiring some specialist equipment and a degree of knowledge about gray partridge husbandry. Modern-day shooting has relied increasingly on the use of rearing and releasing of large numbers of pheasants and red-legged partridges to produce sufficient numbers of birds for shooting. This approach has also been applied to the rearing and releasing of gray partridges. It is easy to obtain large numbers of gray partridges for release by obtaining eggs from captive-reared stock birds, hatching them in incubators and rearing the chicks under electric brooders. Finnish research into the fate of radio-tagged wild and released birds showed that released birds had lower survival and breeding success compared to wild birds (Putaala and Hissa 1998, Putaala et al. 2001). Many of the released birds starved or were predated, and the work concluded that released birds would contribute little to boosting threatened wild populations (Putaala and Hissa 1998). A Scottish study also found that the majority of released birds were predated shortly after being released (Game Conservancy Trust 2000). This method has been used to re-establish wild gray partridge stocks in an area of Northern Italy, where habitat was suitable and it was likely that the gray partridge had succumbed to overshooting. After curtailing shooting and instigating habitat improvement and predator control, thousands of reared gray partridges were released during autumn. Some of these birds survived over winter and bred in the following year (P. Tout, personal communication). The success of this project seemed to be dependent on the release of very large numbers of birds into a well-managed area.

The Edmonthorpe Method

Eggs taken from the wild are hatched and reared by bantams to produce a captive breeding stock. This breeding stock is held as pairs in captivity and any eggs laid by these birds are removed and hatched, usually in an incubator. At two weeks of age the chicks are fostered back to a pair of birds from the captive stock. If more young are produced than pairs are available, then the young are fostered to a single adult bird. At eight weeks of age the chicks and the foster parents are moved to an area of specially prepared habitat and held in captivity for a few days before release. The area of specially prepared habitat consists of strips of maize, a cereal crop, stubble from the previous year and sown grass. The arrangement of these strips allows a high density of coveys to be released. The captive breeding stock is replaced annually by retaining hatched young or hatching eggs from the wild under broody bantams.

This system seems to provide enough birds to shoot and also contributes to the wild stock of partridges.

The François Hughes Method

Having obtained a breeding stock of birds from the wild, this system maintains the birds in captivity in 16-20-m^2 vegetated pens, of which half is mown on a regular basis, the other half left undisturbed to provide breeding cover. In order to keep the captive stock as wild as possible, wild genes are introduced annually by the collection of over-mown wild clutches. The birds are left to pair naturally, produce and hatch eggs and rear their own young. These family coveys are then available for release as a real family group containing two adults and an average of 13 at least 6-week-old young.

This method has supplied birds for restocking in France mainly, but also in Scotland, Ireland and Switzerland (see above).

Translocation outside the natural range

Numerous attempts to introduce the gray partridge to areas outside its natural range have been undertaken at various sites around the world, primarily to provide hunting opportunities (Long 1981). These releases generally involved the releasing of groups of partridges caught from the wild in Europe. The numbers released ranged from groups of less than 100, to in some cases over 20,000 birds. Although it is not recorded, these birds were probably held in enclosures for a few days to allow acclimatisation, before being released en masse.

During the period from about 1880 to the mid-1900s gray partridges were released at numerous locations throughout the United States of America. Some of these releases were highly successful, and today gray partridges are found throughout much of the Mid-west. In Canada, gray partridges were released from about 1900 to the 1930s, and are established in south central Canada today. Other unsuccessful attempts at introducing gray partridges were undertaken in Hawaii, New Zealand, Australia, Fiji and Chile in the late 1800s and early 1900s.

Discussion

Habitat provision and predation control

Underlying the decline of gray partridges throughout its range has been the degradation of feeding and nesting habitats and the increase in predation pressure. Numerous studies have shown that the provision of the correct habitat features can help, in part, to restore gray partridge numbers and this has now been adopted nationally in the UK within agri-environment schemes. A fundamental pre-requisite to any intended gray partridge reintroduction or supplementation is that suitable partridge habitat must constitute a minimum of 6% of the core release area (e.g. Buner et al. 2005). With respect to predation, a study on Salisbury Plain showed that predator control increased wild gray partridge density 2.5 times over three years (Tapper et al. 1996). No study has looked specifically at the effect of predation on released birds, although it is recognised as one of the main factors affecting the survival of released gray partridges. For example, over 90% of gray partridges were dead within six months of releasing compared with 30-70% of wild birds in the same area of the UK, and the highest single factor causing these losses was predation (Dowell 1990*b*). In Finland, where gray partridges were released in both autumn and spring, the losses to predation were between 60-80% (Putaala et al. 2001). Effective predator control is thus also an important pre-requisite to releasing gray partridges, and should be in place before re-establishment begins.

Egg provision

For restocking, every gamekeeper would, without doubt, prefer to have eggs picked up from the wild. It is generally assumed that wild eggs represent true genetic stock, whereas eggs from game farms are somehow genetically impure. It is possible that some of the instinctive behaviour of the birds, including aspects of breeding, foraging and anti-predator behaviour, may be altered by generations of captive breeding. However, whether this behaviour originates from genetics or is acquired by learning has not been established.

Some gamekeepers have set up their own stock of breeding birds from gray partridges caught as adult pairs in the wild and held in captivity or from captive-reared birds hatched from wild eggs and held over-winter. In both situations, it is necessary to ensure that the risk of predation and disease is removed. Gamekeepers have reported that birds caught directly from the wild do not adapt well to captivity, are nervous, lay only small clutches of eggs, tend to lay later than wild birds, and probably as a result of increased stress levels are more susceptible to disease. However, better pen construction and provision of cover may improve performance.

Incubation and rearing

There are three principal methods available to hatch gray partridge eggs. Probably the easiest way is to incubate them in an electric incubator. The resultant young are then usually reared under an artificial heat source, either an electric or gas brooder, but it is possible to foster these chicks, as day-olds, to captive gray partridge adults without chicks or a broody bantam. A second popular option for hatching eggs is under a broody bantam, which is usually left to rear the young. It is possible, however, to take the young away from the bantam and foster them to a captive pair of gray partridges without chicks. The third option is to allow eggs laid by captive birds to be incubated by the birds themselves (parent-rearing). Many keepers (and experience on the GWCT rearing field) have reported that such birds lay 30-50 eggs, are poor incubators, frequently produce infertile eggs and consequently produce few to no chicks. Dowell (1990*c*) showed that only one out of nine pairs caught directly from the wild laid and incubated eggs, whereas gray partridges from a game farm could be induced to undertake incubation. The use of bantams and artificial incubators/brooders is relatively straightforward, although subsequent behaviour may be affected. Dowell (1990*a*) showed that, under experimental conditions, gray partridge chicks reared in captivity by gray partridges respond more appropriately to aerial predators and adult warnings than chicks reared by bantams or artificially. After release, all of the captive-reared chicks, irrespective of the rearing method, suffered greatly reduced survival in the short term (25 days) compared to wild-reared birds, and suffered large over-winter losses (90%) in the longer term. Although based on small sample sizes, there was a suggestion that bantam-reared and adopted (after release) broods fared better than gray partridge and artificially reared young. This has recently been supported by releases in Switzerland (Buner and Schaub 2008), see above.

Young provision

In the absence of a supply of eggs or if there is no desire to hatch eggs oneself, it is possible to acquire day-old chicks from game farms. These could be reared under artificial heat or fostered to a broody bantam or a captive pair of gray partridges.

Establishment of free-living adults

One of the most widely used techniques today for stocking gray partridges using traditional methods involves fostering 6-8-week-old gray partridges to failed wild pairs. However, this is dependent on the presence of resident gray partridge pairs onto which young can be fostered. In situations where gray partridges are completely absent or at very low densities, this technique is not applicable. One solution is to establish a free-living founder population of adult birds, which either breed themselves or most likely can act as foster parents. There are three potential options to achieve this: 1) to rear chicks to about 6-8 weeks and release them either in a single large group or as a number of smaller groups. This mirrors the modern rearing and releasing system used to provide birds for shooting. 2) To release a covey consisting of adults and young birds in autumn. The covey would most likely be made up of a pair of adult captive-reared birds with about 10 young fostered to them. In either case, over-winter losses may be high. To overcome this, a third option would be to hold captive-reared gray partridges over winter and release them as pairs in early spring. The birds would then be left to breed and produce young or, if they failed to breed successfully, to act as foster parents later in the breeding season.

Developing a system for use in gray partridge conservation

A national re-establishment project needs to be cost-effective, relatively easy to undertake and lead to self-sustaining partridge populations. However, sites for re-establishment must be within the species' historical range, and crucially should have at least a minimum of appropriate management in place, including provision of nesting habitat, brood-rearing and over-winter cover, and ideally the presence of a gamekeeper. At least 6% of the core area and 3% of the total area should support habitat suitable for nesting, feeding and cover. There should be no intention to shoot gray partridges until at least 20 birds per 100 ha in the autumn has been achieved and maintained, and measures must be undertaken to ensure that accidental shooting of gray partridges does not occur.

The use of wild-origin birds or eggs for all intended translocations or releases would be ideal. Alternatively, the best system for producing birds for release would use eggs laid in the wild, hatch them under captive gray partridges that would also rear the chicks, then either release the chicks with the adults or foster them to wild gray partridges. However, wild eggs and particularly wild adults are not available in sufficient numbers and it is unlikely that captive-reared gray partridges would make sufficiently good parents to produce enough young. Therefore a compromise is needed to produce a system that is easy, practical and cost-effective, but would produce young gray partridges of sufficient number and quality for re-establishment purposes.

We therefore recommend obtaining eggs from a reliable source, namely a game farm, then hatching the eggs under bantams, which would also rear them to eight weeks. These young would then be fostered to failed wild pairs of gray partridges. A suitable alternative might be to hatch and raise chicks under artificial heat sources and foster these to wild pairs.

If no pairs of wild gray partridges are present it will be necessary first to establish free-living adults. We see two possible methods for doing this, namely release of family groups in autumn or pairs in spring. In order to establish these coveys, gray partridges could be obtained from a game farm in February, held in a large pen and, after pair formation, each pair would be allowed to breed. As it is unlikely that many of these pairs would produce young, day-old chicks would be obtained from a game farm and reared to 4-6 weeks of age. Batches of 13 chicks would be fostered onto each pair to form a covey of 15 birds, for release in late autumn. The second method involves releasing pairs in late March, when the main period of over-winter predation has passed. To establish these pairs, day-old chicks would need to be obtained from a game farm in early summer and held in a large pen until the end of the winter. In early spring these birds would need to be moved into groups of about 50 birds, allowed to form pairs and released shortly after. In both cases, there would need to be a period of acclimatisation to allow them to adapt from pelleted food to natural food such as seed mixtures and grass.

A 2-year research project currently being undertaken by The Game Conservancy Trust will monitor the success of chicks released on 26 study sites in 2 distinct regions of the UK using exactly the techniques outlined above.

Current knowledge gaps and recommendations for future work

This review highlights a number of areas where knowledge is lacking and further research is required. These are highlighted below.

1) It is not known whether young produced from eggs, laid in the wild by truly wild parents, and ones from eggs laid by captive game-farm parents and reared under identical conditions, will behave and survive differently or have different breeding success when released into the wild.

This could be explored by obtaining eggs from the two sources, hatching and rearing them under identical conditions, releasing them into the wild and monitoring their survival and breeding success following release.

2) It is not known whether there are differences in the behaviour and survival of young hatched and reared by either captive gray partridges, a bantam or an electric heat source, then adopted by wild barren pairs.

This could be explored by hatching and rearing young under the three different systems, then releasing the young reared by these three methods and monitoring their subsequent success.

3) It is not known if the release of coveys in autumn or of pairs in spring is a more effective method of re-establishing free-living adult gray partridges.

This should be investigated by releasing both coveys and pairs of gray partridges and monitoring the success.

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How to Re-Introduce Gray Partridges - Conclusions from a Releasing Project in Switzerland

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Once a widespread farmland bird in Switzerland, the Gray Partridge (*Perdix perdix*) has declined drastically since the 1960's as a result of agricultural intensification and changes in predator abundance. In 1991 the wild population had dropped to ≈17 pairs only. We initiated a reintroduction experiment of Gray Partridge to investigate its feasibility for conservation of the species in Switzerland. Between 1998 and 2001 we released 145 partridges in the Swiss Klettgau, an intensively cultivated area from which the species had become extinct in 1993, but had since been ecologically enhanced with wild-flower strips and hedges up to 5.8% of the available habitat in the release area (≈530ha). Although the duration of the study was too short to prove if further partridge re-introductions in Switzerland or abroad will be successful in terms of creating self-sustainable populations, it is possible to draw four basic conclusions for future partridge re-introduction projects: 1) prior to every partridge re-introduction or re-establishment the habitat must be enhanced with permanent habitat structures, 2) if translocated wild birds are not available for release, chicks should be fostered whenever possible to increase their survival, in the best case to wild birds still resident in the area, 3) reintroductions should only be envisaged in areas with low predator numbers and human activities, 4) in order to find possible weaknesses in re-introduction projects, post-release monitoring is essential to ensure the project targets are met.

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Introduction

In the last forty years, the Swiss lowlands have lost much of their former value as habitat for wild plants and animals, mainly because of the effects of modern farming practices. One of the most prominent and charismatic species affected is the Gray Partridge (*Perdix perdix*). Once a typical farmland bird with a spring population in the 1960's of about 10,000 individuals, its numbers have declined sharply since (Schmid et al. 1998). Considering the alarming decline of the Gray Partridge, the Swiss Agency for the Environment, Forests and Landscape (SAEFL) entrusted the Swiss Ornithological Institute in 1991 to undertake a ten year project on "Protection measures for Brown hare and Gray Partridge" (see also Jenny et al. 2002). In this context the "Klettgau" in the canton of Schaffhausen was chosen as study area as it was one of the two regions in Switzerland where wild partridges still remained in small numbers at that time. To reverse the main cause of the partridge's decline - habitat loss (e.g. Potts 1986) - the main activity in the early stages of the project was the promotion of ecologically enhanced habitats such as wild-flower strips and hedges. Unfortunately, the already very small partridge population went extinct shortly after the beginning of the project. However, by 1998 the area of partridge-friendly habitats had grown to such an extent that itallowed us to start a reintroduction experiment. This was undertaken as part of the Swiss Ornithological Institute's project "Birds as test organisms to evaluate enhanced habitat diversity in agricultural areas" which started in 1996. This project became part of the interdisciplinary research program "Integrated Project Biodiversity" launched by the Swiss

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National Science Foundation. The main aims of the re-introduction project were to assess whether Gray Partridges 1) are able to survive and reproduce in an ecologically enhanced landscape such as the Klettgau, 2) make use of the enhanced areas and, if so, which value they have for them, 3) are disturbed by human and predator activities within the re-introduction area.

This paper summarizes the results of the project published elsewhere (Buner et al. 2005, Buner and Schaub 2008, Buner 2006) and concludes on their basis if and under what conditions further partridge re-introductions in Switzerland and abroad might be successful.

Study Area

The study was conducted in the intensively cultivated arable region Klettgau near Schaffhausen, Switzerland (430 m a.s.l.), where mainly cereals (49%), oil-seed rape and sunflowers (14%) and root crops (12%) are grown. Grassland covered 11%, another 11% were bare of vegetation (buildings, roads, etc). Permanent cover such as wild-flower strips, hedgerows and grass banks amounted to 3% of the area. Field sizes ranged from 0.1 to 5.5 ha. The study area was c. 530 ha in size. From 1991 to 2001 the area of wild-flower strips increased from 0 ha to 12 ha, the area of hedgerows from 2 ha to 2.7 ha. Wild-flower strips were narrow strips of 6-20 m width along field edges and were maintained for at least six years. They were initially sown with a mixture of c. 30 wild plant species, however, a total of 234 plant species were recorded in the wild-flower strips in the research area (Ullrich 2001). Besides agricultural use, the study area is very popular for recreation mainly by walkers with or without dogs and riders. In the centre of the study site there is a dog training school and an arena to school horses, in the northern part of the area there is a cycling route. For further description see Buner et al. (2005).

Methods

From 1998 to 2000 we released a total of 142 partridges in the study area, all genetically originating from the western clade of the subspecies *Perdix perdix perdix* (see Liukkonen-Attila et al. 2002). Originally it was planned to release wild translocated birds only (50 per year) as it is well known that this technique generates the highest possible re-introduction success. After the first year of this study however, it became apparent that it was impossible to organize enough wild-caught birds for release (n = 33 released in total). We therefore had to switch to the next most promising option which I considered to be parent-reared birds released as coveys in autumn (n = 77). In the second and third year of this study we additionally fostered parent-reared chicks (n = 32) to already re-established adult birds which had failed to produce their own young. Each radio-tagged bird was located at least once every week until it was found dead. For detailed descriptions of the origin of birds released, releasing techniques, transmitters, data collection and catching techniques used, see Buner et al. (2005) and Buner and Schaub (2008).

Results

Dispersal, survival and causes of death

Of 110 released and radio-tagged adult partridges (33 wild translocated, 77 parent-reared), 73% remained within the study area and 52% survived the first month after release. During the first month after release, they frequently moved across the whole research area. After settling, 98% of all partridge locations were recorded in that part of the study area where the density of enhanced areas was maximal.

Monthly survival was highest in wild-hatched partridges of the founder population (mean ± SE; 0.90 ± 0.03), followed by that of fostered chicks (0.86 ± 0.03) and translocated adult wild birds (0.82 ± 0.06). While survival of these groups was not statistically different from each other, survival of captive-reared adults was significantly lower (0.70 ± 0.06). We found the carcasses of 91 partridges; 88 of them were predated, 1 died because of disease, 1 because of a traffic accident and 1 as a result of a territorial fight. Predation by mammals (mainly foxes) was twice as frequent as predation by avian predators. (For more details see Buner and Schaub 2008).

Reproductive success

We radio-monitored 19 pairs that started egg laying, as all other birds released were either predated before the breeding season started or dispersed (see above). Of those 19 broods, seven hatched and twelve failed (11 predated, 1 disturbed). Mean clutch size of first clutches was 15.3 eggs (n = 9 clutches, SE = 0.27). Only one replacement clutch was found. 86% of all eggs hatched (n = 7 broods, of which an avg. of 6.43 (SE = 1.86) juveniles per brood survived until October). The average percentage of successful nests over three years was only 0.33 (SE = 0.08). When breeding the year after release, fostered chicks tended to have more successful nests (0.44 [SE = 0.43]) than when individuals of the other treatment groups were involved (reared adults: 0.17 [SE = 0.03]; translocated: 0.25 [SE = 0.07]; wild hatched in study area: 0.27 [SE = 0.29]), but the differences were not statistically significant due to low sample sizes (χ^2_3 = 0.68; P = 0.88, Generalized linear mixed model with a binomial error and the brood identity as random factor). Eleven out of 19 nests were located in wild-flower strips.

Habitat use and home range size

At the level of the individual family group (pairs or coveys), we found a significantly greater use (throughout the year) of habitat areas that were enhanced with wild-flower strips and/or hedges, compared to non-enhanced areas. When the birds used the agricultural fields, densities of use declined sharply with increasing distance from the nearest enhanced area. Thus, the availability and spatial distribution of ecologically enhanced areas were the main determinants of the partridges' range use. Despite their strongly over-proportional use of enhanced areas, the partridges spent a large proportion of time in cultivated fields. In summer, frequently visited vegetation types were cereals (average 26.1% of locations), root crops (14.8%) and grassland (9.3%). In winter, the birds spent much of their activity in cereals or stubble fields (32.7%) and rape (24.1%). This indicates that these types of vegetation, particularly cereals, were attractive resources, although not preferred in respect to their availability.

The size of the group home-ranges varied significantly with season. In spring (pre-breeding period) and summer (breeding period), the average home-ranges (± SD) were 6.8 (± 4.0) ha and 6.9 (± 2.6) ha, respectively. From late summer until the end of winter (non-breeding period), the home-ranges were significantly larger (late summer: 15.2 (± 6.6) ha; autumn: 17.0 ± (4.0) ha; winter: 14.4 (± 3.6) ha). For more details, see Buner et al. (2005).

Disturbance

Partridges showed a distinctive cause-specific reaction repertoire to all disturbance types compared, mainly crouching in presence of raptors and showing vigilance in presence of mammals (foxes and cats). Flushing was the main reaction when disturbed by leisure activities. When flushed, partridges reduced their flight distance by 54 metres compared to unforced flights and remained in their territory in 87% of all cases. In summer, their main escape cover was cultivated fields, whereas in winter they mainly used permanent cover such as wild-flower strips and hedges. The spatial distribution of partridges was influenced by season: In summer, partridges avoided areas with high human disturbance, whereas in winter they avoided areas with high predator abundance and close proximity to tall hedges. Human activities caused twice as much disturbance events as predators, with associated energetic costs. Overall, disturbance substantially limited overall spatial use, with consequences for the carrying capacity of the area.

Discussion

Despite the best efforts of Swiss agricultural policy and millions of Swiss Francs spent on enhancing agricultural biodiversity, the Gray Partridge, a key farmland bird species, has shown no sign of recovery to the present day. Indeed, the last truly wild partridge population in Switzerland in the Champagne genevoise became almost extinct during the time of this study (3 birds left in 2004). Much more effort is needed therefore, to save the Gray Partridge which depends on high quality habitat enhancement in the right places and in substantial proportion of the available habitat supply (for more details see Buner et al. 2005). To save species like the Gray Partridge, carefully planned projects which bring together various stakeholders such as farmers, population biologists, conservationists, game keepers, people from the local, regional and national government, local nature conservation groups and the press are necessary. The results of this re-introduction project in the Klettgau show that with enough staying power, even the intensively exploited Swiss countryside may provide a suitable environment for highly demanding species such as the Gray Partridge. Even though it is not possible to prove from this study's results whether further partridge re-introductions in Switzerland or abroad will be successful in terms of creating self-sustainable populations (to do so, long term experiments with more birds involved are necessary) it is possible to draw four basic conclusions for Gray Partridge re-introduction projects:

1. Prior to every partridge re-introduction or re-establishment project the habitat must be enhanced with permanent habitat structures. Wild-flower strips and low, if possible treeless hedges are highly preferred by partridges as they provide nesting, brood rearing, foraging and escape cover during all seasons.

2. If translocated wild birds are not available, the most efficient releasing technique is fostering chicks to pairs which failed to hatch their own young. In the best case, chicks are fostered to wild birds still resident in the area. Where no such birds are left, captive parent-reared adults should be released as coveys in autumn with maximum support to allow successful settlement, followed by fostering chicks the following summer. Giving the system enough time to develop, a carefully planned releasing regime should allow a population of well experienced individuals to establish in a relatively short time.

3. To enhance the chances of re-introduction success, areas should be chosen with low predator numbers and human activities, especially leisure activities, or managed specifically to reduce those sources of disturbance. Predators and human activities do not only have direct impacts on survival and breeding success but may also reduce the available area for foraging and therefore the carrying capacity of an area as a whole.

4. Sustained post-release monitoring should check for winter mortality, nesting success and chick survival over time. At least one spring and autumn count should be carried out to assess population development of the released birds. In order to find possible weaknesses in a re-introduction project, knowledge of the most important population parameters are essential in order to ensure the project targets are met.

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Comparison of Survival, Productivity, Movements, and Habitat Use of Pre-Season Released Quail on Wild Northern Bobwhites on Groton Plantation, South Carolina

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To understand the effects of a pre-season release operation (liberating pen-raised quail for hunter harvest) on native quail (*Colinus virginianus*) populations, we developed studies to assess population demographics and movements of three treatment groups (wild, pen, and wild/pen quail) on Groton Plantation, Allendale County, South Carolina. Two isolated study sites were selected: a site with only wild quail (control) and a site with both wild quail and pen-raised quail released each September (treatment). We radio-tagged wild (n = 306) and pen-raised (n = 330) quail during 1996 and 1997, for monitoring various demographic parameters, including body weight, survival rate, home range, habitat use, linear dispersal, and hunting susceptibility. Based on data from this study, the release of pen-raised quail affected the behavioral characteristics of wild quail and possibly physical characteristics such as body weight. Individual body weight measurements indicated that a higher percentage of wild/pen quail weighing more than wild quail during both years (March 1996 and 1997). While these data are not direct measurements of introgression, reproductive success of pen-raised quail was observed during two consecutive breeding seasons (44% and 22%). Behavioral characteristics such as home range size, habitat-use, and linear dispersal were different between wild/pen and wild control quail during specific periods (i.e., season and year) potentially causing lower survival rates (0.077 $\pm$ 0.074) during the 1997 overwinter season and increased hunter susceptibility. While the pre-season release of pen-raised quail can produce economically efficient hunting, negative impacts on native quail population may occur.

Citation: Eggert DA, Mueller BS, Robinette L, Wellendorf SD. 2009. Comparison of survival, productivity, movements, and habitat use of pre-season released quail on wild northern bobwhites on Groton Plantation, South Carolina. Pages 396 - 408 in Cederbaum SB, Faircloth BC, Terhune TM, Thompson JJ, Carroll JP, eds. Gamebird 2006: Quail VI and Perdix XII. 31 May - 4 June 2006. Warnell School of Forestry and Natural Resources, Athens, GA, USA.

Key words: *Colinus virginianus*, northern bobwhite, pen-raised quail, pre-season release, South Carolina

Introduction

The pre-season release of pen-raised northern bobwhite quail (hereafter: quail) has become a common management practice for landowners interested in augmenting quail populations for hunting. Typically, in a pre-season release operation, pen-raised quail are released two months prior to the start of hunting season, which may provide them to acclimation to their new environment and develop suitable flight characteristics more similar to their wild counterparts. DeVos and Speake (1995) suggested that pen-raised quail could acquire wild characteristics such as sustained holding times before flushing, increased flight intervals, social continuity as a flushing covey, and similar summer survival rates.

Many concerns regarding the ecological impacts of releasing pen-raised quail on wild quail have been postulated. Two problems proposed are a reduction is survival due to increased mortality from predators and reduced fitness of native quail (Landers et al. 1991). DeVos and Speake (1995) have documented that pen-raised quail interact within wild coveys, and presumably are able to nest successfully; however, the latter lacks empirical support. Most research involving pen-raised quail has focused solely on their fate and dispersal among varying habitat

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conditions, and not on their interactions with native quail populations or possible reproductive contribution.

Our research objective was to determine the effects of pre-season released quail on the demographics, habitat use, and dispersal of a native quail population in the coastal region of South Carolina. To accurately assess these impacts we used a site managed strictly for wild quail as our experimental control. We quantified body weight, survival rates, cause-specific mortality, hunter harvest, linear dispersal, home range, and reproductive success.

Study Area

Research was conducted on Groton Plantation located in the lower coastal plain of Allendale County, South Carolina adjacent to the Savannah River. Two study sites were selected, a pre-season release area (PR) and a control site (WC) managed strictly for wild quail. The linear distance between the two sites was approximately 2.4 km to minimize quail movements between study sites. Both sites were similarly managed to promote quality quail habitat. The PR site (789 ha) was comprised of upland forests (38%), field systems (34%), transitional buffers (22%) and lowland areas (6%). The WC site (406 ha) was comprised of field systems (43%), upland forests (31%), transitional buffers (21%) and lowland area (5%).

Field systems were divided into two major components: disked fields and fallow hedgerows. Fallow hedgerows comprised roughly two-thirds of the field system, whereas disked fields accounted for the remaining one-third. Disked fields were maintained by annual fall and mid-winter harrowing. Fallow hedgerows were border areas within the field system, established for protective refuge (escape cover) and traveling corridors for quail. Upland forest habitat consisted mostly of mixed pine/hardwoods (*Pinus* spp. and *Quercus* spp.) with an average stand density of 6 - 13 m^2 /hectares. Approximately 80% of each site was annually burned during winter. Lowland sites were typically defined as bottomland hardwoods, which regularly flooded or had high soil saturation characteristics and scarce ground story vegetation. Transitional buffers were defined as a 20 m interval zone between field systems and upland forests (10 m into each habitat type from the juxtaposition).

Methods

Trapping

Quail were captured using night netting and standard walk-in funnel traps (Stoddard 1931) baited with sorghum. Each trap site was pre-baited for approximately one month and pre-season release sites were supplementally fed throughout the hunting season for both study years. The wild control site was supplementally fed only during the second study year. Quail were aged (Rosene 1969), sexed, weighed, banded, and fitted with a 6 g necklace transmitter (American Wildlife Enterprises, Monticello, FL 32344) if their weight exceeded 130 g. Transmitters had mortality indicators that were activated after a motionless period of 12 hours. Trapping was conducted until at least 30 radio-tagged quail had survived greater than 1 week on each treatment site.

Quail were classified into three treatment groups: wild, wild/pen, and pen-raised quail. Native quail captured from the WC site were referred to as "wild", whereas unbanded native quail captured from the PR site were referred to as "wild/pen." Pen-raised quail, referred to as "pen" were leg banded prior to release for identification and were either trapped or released on the PR study site. Research study was conducted from March 1996 to March 1998 and years were divided into two distinct seasons, breeding season (Mar-Sep) and overwinter season (Sep-Mar) within each year.

Telemetry

Radio-tagged quail were monitored using homing techniques (White and Garrott 1990) with a 2-element "H" antenna and receiver. All birds were tracked $\geq$3 days per week and locations were recorded on maps generated with AtlasGIS. Each location was assigned UTM coordinates and a habitat-use classification such as field systems, uplands,

Table 1: Trapping weight results analyzed for treatment groups (pen, wild/pen, and wild) by season and year on Groton Plantation, Allendale County, SC.

Trapping Period	Trt Group	n	Weight (g)	SE	Quail $>$183.3g
Mar-96	Pen[a]	64	206.3	1.875	64
Mar-97	Pen[c]	74	201.2	1.744	66
Mar-96	Wild/Pen	34	177.9	2.573	13
Mar-97	Wild/Pen	49	171.9	2.122	11
Mar-96	Wild	54	161.9	2.042	2
Mar-97	Wild	43	172.3	2.288	2
Sep-96	Pen[b]	97	175.5	1.523	25
Sep-97	Pen[b]	104	190.4	1.471	56
Sep-96	Wild/Pen	41	153.6	2.343	1
Sep-97	Wild/Pen	29	186.7	2.786	15
Sep-96	Wild	43	144	2.288	0
Sep-97	Wild	40	159.8	2.372	2

[a] Trapped pen-raised quail that were released on or before Sept 22, 1995, [b] 10 to 12-week-old, pen-raised quail which have not yet been released into the wild, [c] Trapped pen-raised quail that were released on or before Sept 24, 1996.

transitional buffers, and lowland areas. Female quail were tracked once per day throughout the breeding season. Upon successful hatching, two daily locations were recorded until chicks reached flight stage (approximately 2 weeks after hatch). Nesting data such as clutch size, percent catch, nest predation and nest abandonment were also collected during this period. We left-censored quail that moved off the study site, had radio-transmitter failure, or died within the first week of radio-transmitter attachment due to acclimation stress and possible predator induced emigration at release sites (Pollock et al. 1989*b*, Curtis et al. 1989). Cause specific mortality was categorized into four predator classes (i.e. avian, mammalian, hunter, and reptilian) by studying the post-mortem remains, condition of the transmitter (teeth or beak marks), and other evidence at the kill site (tracks, avian whitewash, feathers, bones, etc.; Curtis et al. 1989).

Hunter Harvest

Substantiated efforts to minimize harvest effects were incorporated in management plans on both study sites by extending time between hunts (designated hunting sites were only hunted once every 14 days), allowing no more than 3 birds to be harvested on any given covey, and equalizing hunting pressure throughout the hunting season to avoid overharvest during late-winter.

Statistical Analysis

$$\bar{x} \pm t * \left(\frac{\sigma}{\sqrt{1+\frac{1}{n}}}\right)$$

Body Weights - Wild quail weights on pre-season release sites (wild/pen) were examined for possible introgression effects by calculating the upper 95% confidence interval for individual wild quail weights (control quail from WC) and testing the difference in probability of capturing a wild or wild/pen quail weighing greater than the 95% CI value by chi-square analysis (PROC FREQ; SAS Institute, Inc. 1989). Only weight data from the March trapping periods were used in these analyses since

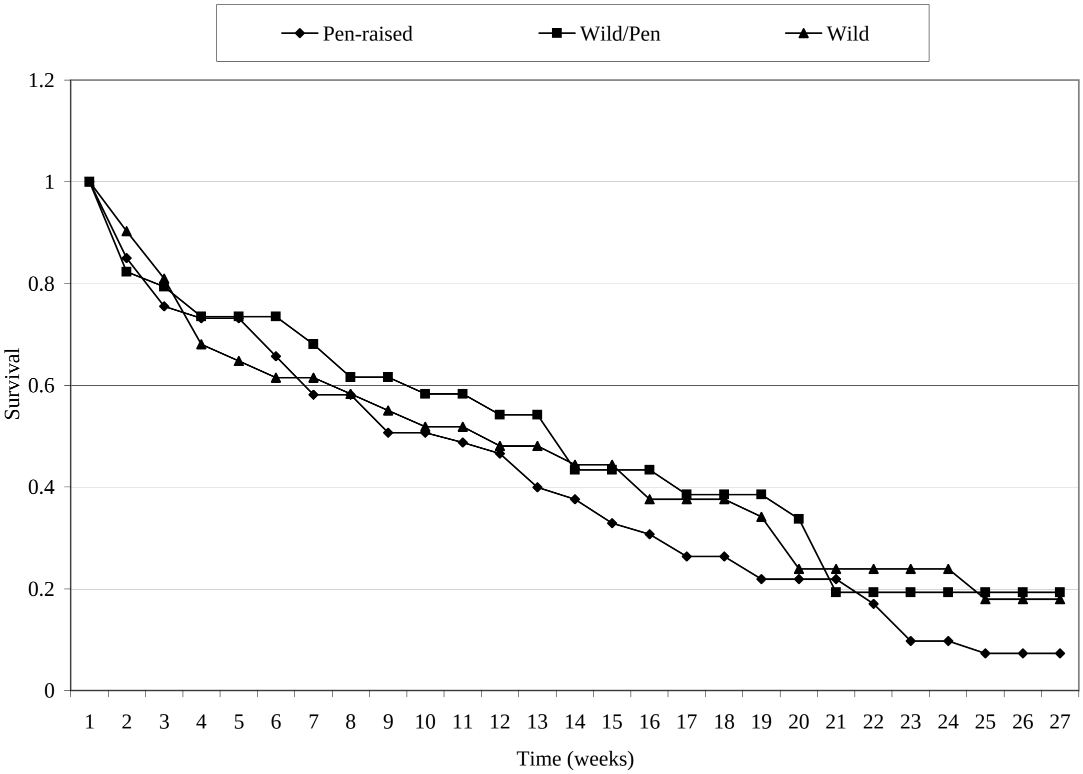


Figure 1: Estimated survival rates of treatment groups (pen, wild/pen, and wild) using a Kaplain-Meier Staggered Entry Design for breeding season year one (18-March to 22-September 1996).

juvenile quail may have negatively influenced the results.

Survival Rates - The Kaplan-Meier staggered entry design (Kaplan and Meier 1958) modified by Pollock et al. (1989*a*) was used to determine weekly survival rate of each treatment group (pen, wild/pen, and wild). Differences in weekly survival rates between treatment groups were tested using a modified normality test (Pollock et al. 1989*a*). The log rank test modified by Pollock et al. (1989*b*) was used to test for differences in survival rates among seasons (overwinter or breeding) and between treatment groups.

Cause-Specific Mortality - Chi-square analysis (PROC FREQ; SAS Institute, Inc. 1989) was used to determine differences in cause-specific mortality between treatment groups (pen, wild/pen, and wild) by year and between years for each treatment group. Reptilian predation was excluded from the analyses due to low frequency of occurrence.

Home Range - Data were analyzed for each treatment group (pen, wild/pen, and wild) by season (overwinter and breeding) and year (1996 and 1997). A minimum number of thirty locations per quail are suggested for home range analysis (Brewer and Fagerstone 1998); however, due to the temporal tracking period (>38 days) and sample size requirements, individual quail with ≥20 locations were used in the analysis. Home range sizes were estimated using 95% adaptive kernel isopleths (Worton 1989), which were generated using ArcView 3.2 (Home Range Extension Program (HRE); ESRI 1989, Rodgers and Carr 1998). The smoothing parameter was estimated using least squared cross validation (LSCV; Worton 1989). The selection of this home range estimator and smoothing parameter was based on the non-normally of the data sets (Seaman and Powell 1996). Due to non-normally distributed home range data for all treatment groups (PROC UNIVARIATE; SAS Institute, Inc. 1989), a logarithmic scale was applied to all home range estimates. Log mean home ranges were compared among treatment groups by year using a one-way analysis of variance (ANOVA, PROC GLM; SAS Institute, Inc. 1989) with LSD as the mean separator.

Habitat-Use - Habitat use data were analyzed using a two-step procedure described by Neu et al. (1974). Chi-square analysis was used to test the null hypothesis that habitat-use was proportional for all habitat types and the z-Bonferroni test was used to determine selection or avoidance of a particular habitat type by calculating a 95% confidence interval for expected use and comparing the values to the observed availability (total hectares of habitat type).

Linear Dispersal - Linear dispersal distance was defined as the farthest linear distance between a quail's release/trap site and observed location. Linear dispersal distances were analyzed for treatment groups by season and year using a one-way ANOVA (PROC GLM) and LSD as the mean separator.

Hunting Harvest On Pre-Season Release Sites - Chi-square analysis was used to test for differences in ratios of wild to pen-raised quail harvested per year (1996-97, 1997-98, and 1998-99) on all pre-season release sites.

Nesting/Reproductive Success - Statistical analysis was not conducted on this data set due to low sample sizes (n < 13), but results are presented in Table 3. An alpha level of 0.05 was established as the significant threshold for all statistical analyses conducted in this study.

Results

Body Weights

We weighed 152 quail in March 1996 and 166 quail in March 1997 (Table 1). The average weight was 206.3 ± 11.3g for pen-raised (n = 64), 177.9 ± 16.8g for wild/pen (n = 34), and 161.9 ± 10.6g for wild quail (n = 54) in March 1996, and 201.1 ± 14.6g for pen-raised (n = 74), 171.8 ± 20.8g for wild/pen (n = 49), and 172.3 ± 7.6g for wild quail (n = 43) in March 1997 (Table 1). To assess possible introgression effects, upper confidence intervals (95% C.I.) for wild quail were calculated as 183.3g for March 1996 and 187.5g for March 1997. Wild/pen quail had a greater probability of weighting more than the upper 95% CI for wild quail during both the 1996 trapping period (38.2%; $P = 0.0004$) and 1997 (22.5%; $P = 0.0145$). During these trapping periods, only 3.7% (1996) and 4.7% (1997) of wild quail weighted more than the upper 95% CI (Table 1).

Survival Rates

Groups By Season - For the 1996 breeding season, estimated survival of wild/pen quail (0.193 ± 0.023, mean ± SE) was greater than wild (0.179 ± 0.081) and pen-raised quail (0.73 ± 0.023), but not statistically different ($P = 0.3576$ and $P = 0.2112$; Figure 1). Similar to the 1996 breeding season, estimated survival of wild/pen quail (0.105 ± 0.019) was greater than wild (0.069 ± 0.022) and pen-raised quail (0.030 ± 0.029), but not statistically different ($P = 0.5092$ and $P = 0.1556$) during the 1997 breeding season (Figure 2). Differences in estimated survival by weekly comparisons were only observed during the latter breeding season between pen-raised quail and both wild/pen (weeks 5 thru 25 except week 9; $P < 0.002$ to $P = 0.0424$) and wild quail (weeks 4 thru 23; $P < 0.002$ to $P = 0.0208$). For the 1996 overwinter season, estimated survival of wild quail (0.594 ± 0.027) was greater than wild/pen (0.417 ± 0.053) and pen-raised quail (0.115 ± 0.108), but only the comparison between wild and pen-raised quail was statistically different ($P < 0.002$; Figures 3). Differences in estimated survival by weekly comparisons for the 1996 overwinter season were only observed between pen-raised quail and both wild/pen (weeks 9 thru 20 and week 24; $P < 0.002$ to $P = 0.0366$) and wild quail (weeks 9, 11 and 14 thru 25; $P < 0.002$ to $P = 0.0434$). During the 1997 overwinter season, estimated survival of wild quail (0.448 ± 0.025) was statistically greater than pen-raised (0.152 ± 0.070; $P = 0.0292$) and wild/pen quail (0.077 ± 0.074; $P < 0.002$). Differences in estimated survival by weekly comparisons for the 1997 overwinter season were only observed

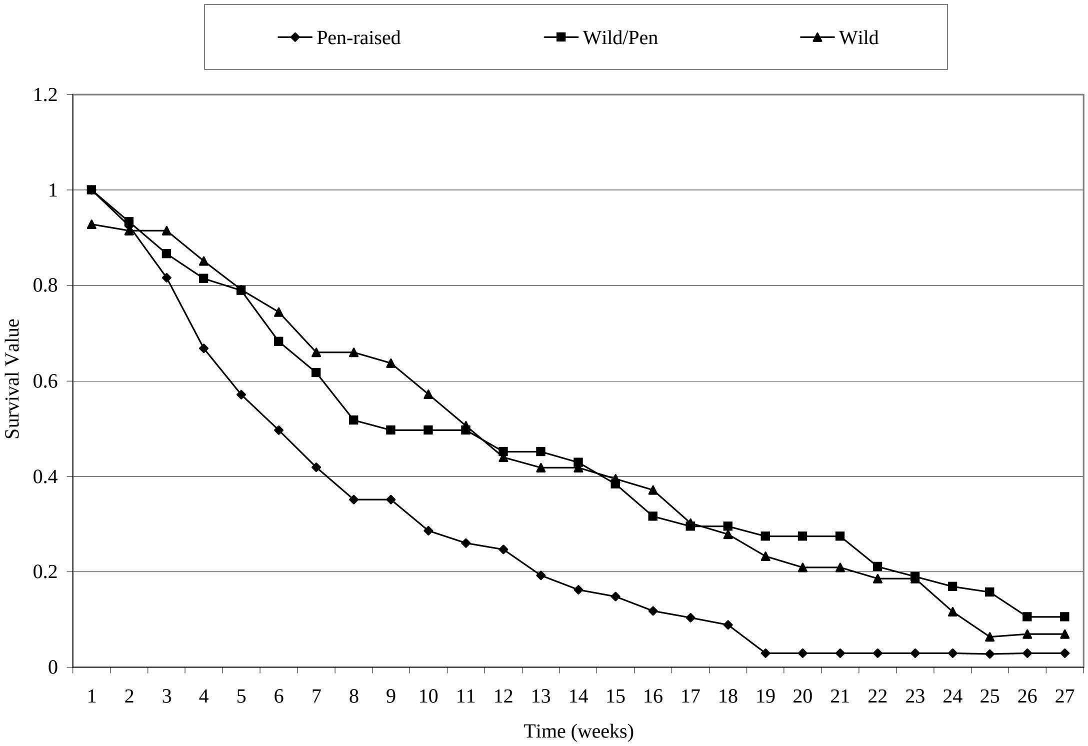


Figure 2: Estimated survival rates of treatment groups (pen, wild/pen, and wild) using a Kaplain-Meier Staggered Entry Design for breeding season year two (10-March to 14-September 1997).

between wild quail and both wild/pen (weeks 9 thru 15, 16, 20, and weeks 22 thru 25; $P < 0.002$ to $P = 0.0324$) and pen-raised quail (all weeks; $P < 0.002$ to $P = 0.0292$). The estimated survival of pen-raised birds from time of release to the start of the hunting season was 0.608 ± 0.008 for 1996 and 0.589 ± 0.006 for 1997 (Figures 3 and 4).

Between Years - For the breeding season, only the data sets of pen-raised quail showed significant differences between years in survival curves (χ^2 = 7.718, df = 1, $P < 0.05$). For the overwinter season, only the data sets of wild/pen quail showed significant differences between years in survival curves (χ^2 = 3.995, df = 1, $P < 0.05$).

Cause-Specific Mortality

We identified cause-specific mortalities of 455 radio-tagged quail by predator class (avian, mammalian, reptilian, and harvest). Avian predators accounted for 51.7% of all mortalities (54% to wild/pen, 51% to wild and 51% to pen-raised quail), followed by mammalian predators (26.3%), hunter (20.3%) and reptiles (1.7%). No statistical differences were observed among treatment groups by predator class; however, mammalian predation increased during the second study year for the all treatment groups (pen 19 to 36%, wild/pen 14 to 26%, and wild 18 to 34%).

Home Range

Pen-raised quail ($\bar{x} = 2.99 \pm 0.124$, $\bar{x} = 2.95 \pm 0.155$, mean ± SE) had smaller log mean home ranges than wild quail ($\bar{x} = 3.52 \pm 0.174$, $\bar{x} = 3.44 \pm 0.141$) during the 1996 and 1997 breeding seasons (P = 0.0124 and P = 0.0199). Additionally, pen-raised quail ($\bar{x}$ = $1.75 \pm 0.093, \bar{x} = 1.73 \pm 0.088$) had smaller log mean home ranges than wild quail ($\bar{x} = 2.81 \pm 0.141, \bar{x}$ = 2.32 ± 0.134) during the 1996 and 1997 overwinter seasons ($P < 0.0001$ and P = 0.0003). Pen-raised quail had a smaller log mean home range

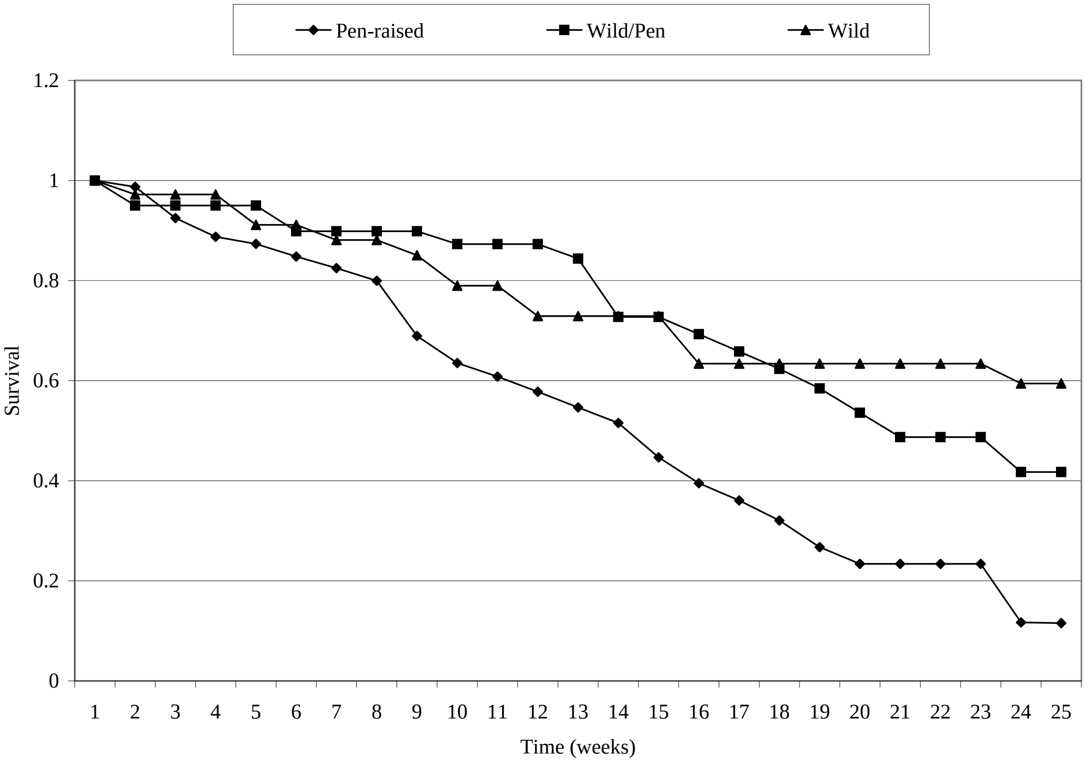


Figure 3: Estimated survival rates of treatment groups (pen, wild/pen, and wild) using a Kaplain-Meier Staggered Entry Design for overwinter season year one (16-September 1996 to 9-March 1997).

than wild/pen quail ($\bar{x}$ = 2.41 ± 0.132) for only the 1996 overwinter season ($P < 0.001$). After logarithmic adjustments, wild quail had significantly larger home range estimates than wild/pen quail for breeding ($\bar{x}$ = 3.52 ± 0.174 vs $\bar{x}$ = 2.93 ± 0.146, P = 0.0099) and overwinter seasons ($\bar{x}$ = 2.82 ± 0.141 vs $\bar{x}$ = 2.40 ± 0.0.132, P = 0.0346) in 1996, but were similar for both seasons in 1997 (P = 0.2457 and P = 0.1733). Logarithmic mean home range estimates were, however, larger for wild than wild/pen quail during the breeding and overwinter seasons ($\bar{x}$ = 3.44 ± 0.141 vs $\bar{x} = 3.20 \pm 0.146$ and $\bar{x} = 2.32 \pm 0.134$ vs $\bar{x} = 2.01 \pm 0.184$) during 1997. Estimated mean home ranges (hectares) of treatment groups by season are presented in Figure 5.

Habitat-Use

Wild quail selected for forested habitat types, used buffers in proportion to the total hectares available, and avoided for both field systems and lowland habitat types (Table 2). Wild/pen and pen-raised quail did not differ in habitat selection; both groups avoided forested and lowland habitat types while selecting for field types and used buffers in proportion to the total hectares available.

Linear Dispersal

Wild quail had the largest linear dispersal distances compared to wild/pen and pen-raised quail among all seasons and years (Figure 6). For the 1996 breeding season, linear dispersal measurements were greater for wild quail (824.2 m ± 75.7, mean ± SE) compared to wild/pen (635.9 m ± 63.5, P = 0.0469) and pen-raised quail (596.1 m ± 55.7, P = 0.0159); however, no differences occurred during the 1997 breeding season (P = 0.0773 and P = 0.4723). For the 1996 overwinter season, linear dispersal measurements were greater for wild quail

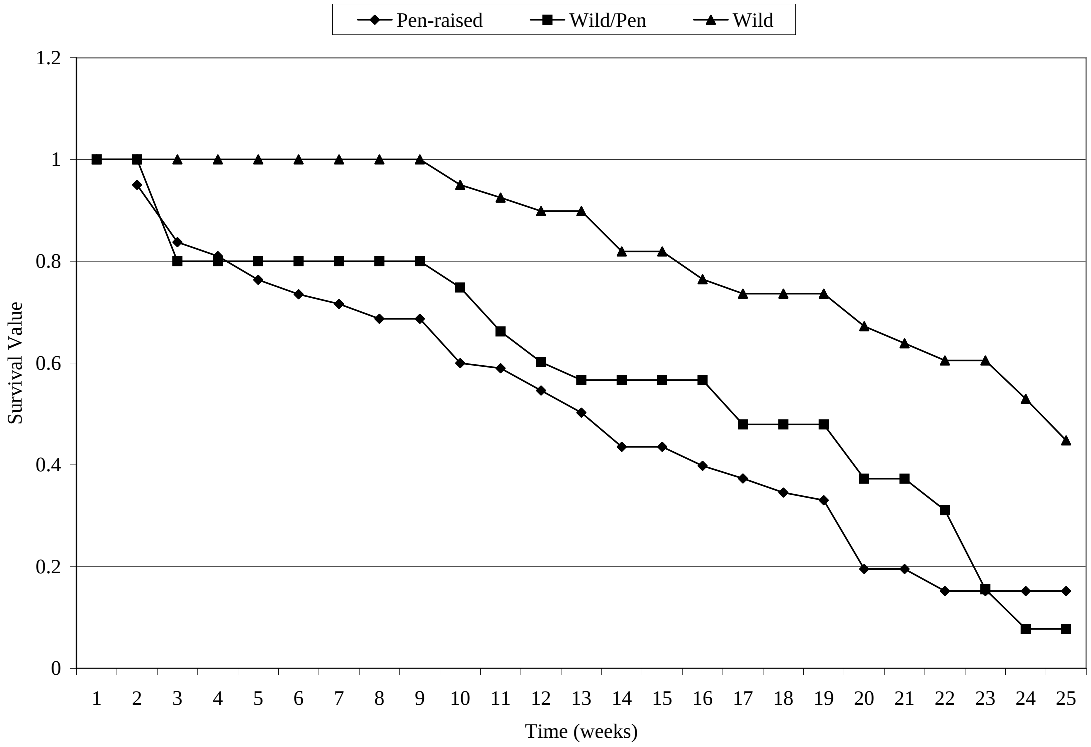


Figure 4: Estimated survival rates of treatment groups (pen, wild/pen, and wild) using a Kaplain-Meier Staggered Entry Design for overwinter season year two (15-September 1997 to March 1998).

(786.5 m ± 90.2) compared to wild/pen (519.9 m ± 85.2, P = 0.0379) and pen-raised quail (416.4 m ± 53.5, P = 0.0008); however, no differences occurred during the 1997 overwinter season (P = 0.4688 and P = 0.5227).

Nesting/Reproductive Success

The ratio of nests located to successful nests decreased during the second study year (1997) for all treatment groups (Table 3). Number of nests predated increased during the second breeding season for pen-raised (33 to 55%), wild/pen (16.7 to 31.3%), and wild quail (30.0 to 41.7%) as well as nesting bird predation (Table 3).

Hunting Harvest On Pre-Season Release Sites

The ratio of wild to pen-raised quail harvested per year was collected from PR and nine other pre-season release sites during 1996-97 (n = 133), 1997-98 (n = 92) and 1998-99 (n = 121; Table 4). This ratio significantly decreased during consecutive hunting seasons including 1996-97 (0.191) and 1997-98 (0.109, $P<0.0001$) and 1997-98 and 1998-99 (0.059, P = 0.0035).

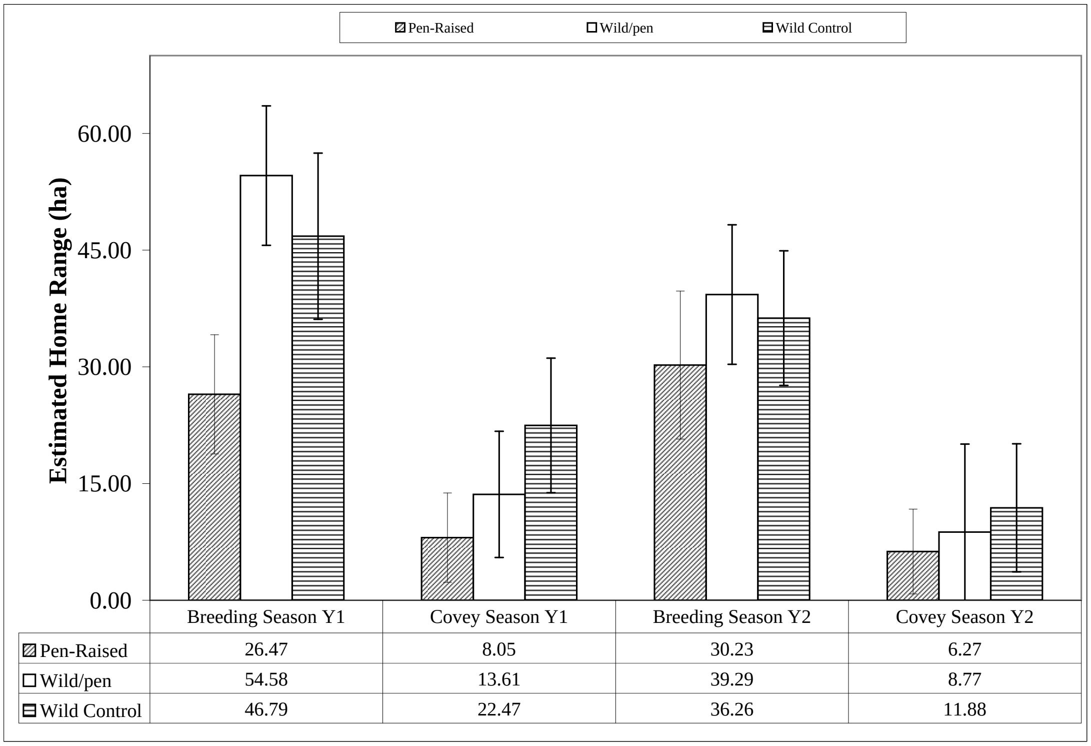
Discussion

Body Weights

Individual body weights of treatment groups were analyzed to test the hypothesis that wild quail on a pre-season release site (PR) have weights greater than native wild quail located on a wild control site (WC). Results from the March trapping periods seemed to support that introgression effects (genetic mixing) might have occurred on PR, but conclusive data is still warranted to assess genetic changes in wild quail populations on pre-season release sites.

Survival Rates

DeVos and Speake (1995) and Sisson et al. (2000*a*) reported higher autumn survival rates for wild quail



	Breeding Season Y1	Covey Season Y1	Breeding Season Y2	Covey Season Y2
▨ Pen-Raised	26.47	8.05	30.23	6.27
☐ Wild/pen	54.58	13.61	39.29	8.77
⊟ Wild Control	46.79	22.47	36.26	11.88

Figure 5: Home range (hectares) estimated using 95% adaptive kernel isopleths for treatment groups on Groton Plantation, Allendale, SC during two seasonal groups (overwinter and breeding season) for both study years. Error bars represent the standard error of the treatment mean.

on a control site compared to wild quail on a pre-season release site. Our results were similar to these studies, in that wild quail had higher survival rates (0.521) during overwinter seasons (pooled) compared to wild/pen (0.247) and pen-raised quail (0.133). Higher wild/pen survival rates for the 1996 versus 1997 overwinter season (0.417 and 0.077) may have been due to a time-dependent relationship between the release of pen-raised quail and various factors that may include predation dynamics (e.g. emigration, changes in prey selection, capture efficiency, and hunter susceptibility) and covey structure and function. Mammalian predation was not significantly greater between the 1996 and 1997 study years, but might have attributed to theses findings as well as influences (Type II errors) caused by low sample sizes.

Cause-Specific Mortality

Avian species have been reported to cause the majority of predation against both liberated and native quail in the southeastern U.S. (Sisson et al. 2000*a*, DeVos and Speake 1995, Curtis et al. 1989). Avian species accounted for 63% of all identified mortalities within our study compared to 69% as reported by Sisson et al. (2000*b*), 86% by DeVos and Speake (1995) and 60% by Curtis et al. (1989). Mammalian predators accounted for the second most identified mortalities (33%), which is similar to values reported by Burger et al. (1995) and DeVos and Speake (1995). Statistical differences in cause-specific mortalities among quail groups did not occur in our study or in the studies of DeVos and Speake (1995) and Curtis et al. (1989).

Table 2: Habitat-use analysis using the Neu et al. (1974) procedure for quail treatment groups on Groton Plantation, Allendale County, South Carolina (18 March 1996 to 8 March 1998).

Treatment Group	Habitat Type	Total Loc.	Expected Use (P)	Observed Use (Pi)	Bonferroni[a] Confidence Int.	Result
Pen	Field systems	4294	0.339	0.532	0.518 = P = 0.545	Preference
Pen	Buffers	1812	0.224	0.224	0.213 = P = 0.236	Proportion
Pen	Forest	1973	0.378	0.244	0.232 = P = 0.256	Avoidance
Pen	Lowland	0	0.059	0	0 = P = 0	Avoidance
Wild/Pen	Field systems	2179	0.339	0.486	0.467 = P = 0.504	Preference
Wild/Pen	Buffers	1016	0.224	0.226	0.211 = P = 0.242	Proportion
Wild/Pen	Forest	1293	0.378	0.288	0.271 = P = 0.305	Avoidance
Wild/Pen	Lowland	0	0.059	0	0 = P = 0	Avoidance
Wild	Field systems	1811	0.431	0.308	0.293 = P = 0.324	Avoidance
Wild	Buffers	1230	0.209	0.21	0.196 = P = 0.223	Proportion
Wild	Forest	2830	0.31	0.482	0.466 = P = 0.498	Preference
Wild	Lowland	0	0.051	0	0 = P = 0	Avoidance

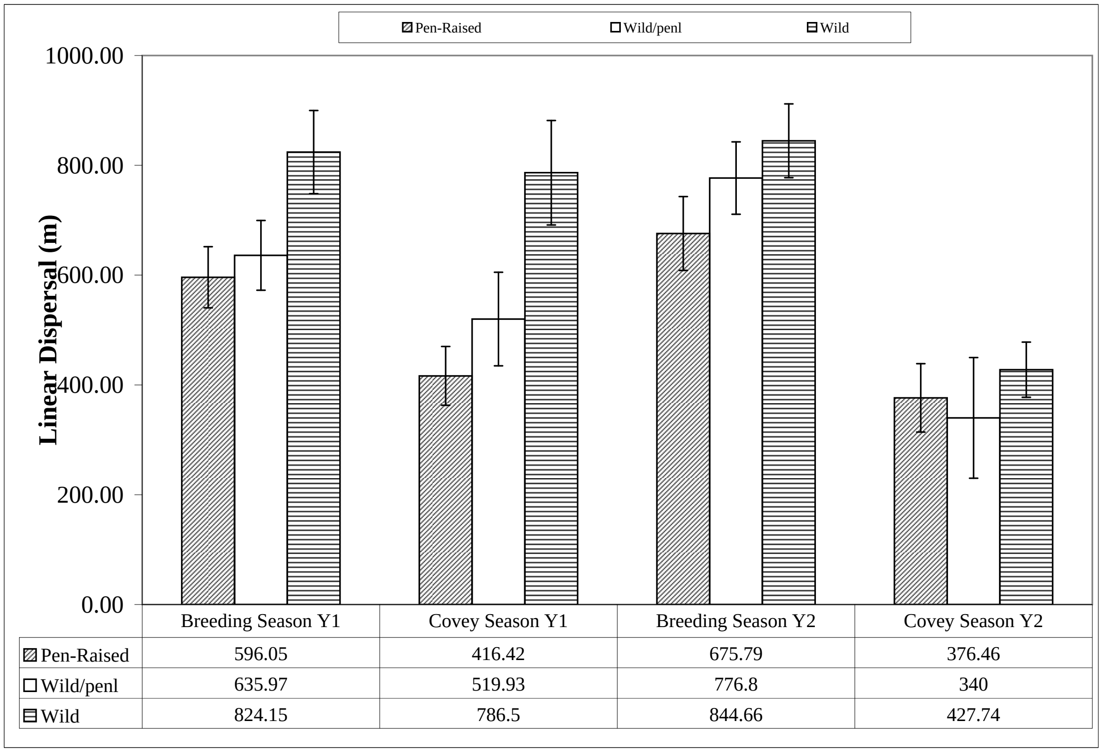
[a] Z statistical test used in Neu et al. (1974) utilization-availability procedure with $\alpha = 0.05$.

Home Range And Habitat-Use

Pen-raised and wild/pen quail did not differ in estimated home range size for all seasonal periods, except 1996 overwinter season, possibly due to the lower percentage of mixed coveys (i.e. covey comprised of both one radio-tagged pen-raised or wild/pen quail) reported between years (33% vs. 36%). These percentages, however, may not truly indicate total mixing of coveys because of the low sample size of radio-tagged pen-raised quail in comparison to the total release (80 to $\sim$8700 pen-raised quail). An important trend that we observed during night-netting attempts was that no "purely" wild covey was captured on PR throughout the entire study period. Wild/pen and pen-raised quail habitat-use analysis resulted in similar selection/avoidance patterns between treatment groups, indicating that home range size may be related to habitat-utilization. Supplemental feeding on PR may have decreased wild/pen and pen-raised quail home ranges during 1996 overwinter season; however, larger home range sizes were reported for wild quail during 1997 overwinter season when feeding practices were equalized between study sites. The lack of significance in home range sizes for 1997 overwinter season (wild/pen vs. wild quail) may have been affected by the loss of tracking data from September to October. Behavioral characteristics such as movement and habitat selection within a covey are not fully understood; however, pen-raised quail seem to have altered these characteristics for the tested wild/pen quail population.

Linear Dispersal

Pen-raised quail appeared to be more stable in the utilization of their summer habitat compared to wild quail and did very little "range shifting" throughout the breeding season. Linear distances (pooled) were the smallest for pen-raised quail during both the breeding season ($\bar{x}$ = 628.4 m) and overwinter season ($\bar{x}$ = 399.4 m). Wild/pen quail had linear dispersal distances more similar to pen-raised



	Breeding Season Y1	Covey Season Y1	Breeding Season Y2	Covey Season Y2
▨ Pen-Raised	596.05	416.42	675.79	376.46
☐ Wild/penl	635.97	519.93	776.8	340
⊟ Wild	824.15	786.5	844.66	427.74

Figure 6: Linear dispersal distances (meters) for treatment groups on Groton Plantation, Allendale, SC during two seasonal groups (overwinter and breeding season) for both study years. Error bars represent the standard error of the treatment mean.

than wild quail, which may be due to their social interactions with pen-raised quail. No pen-raised quail were observed to transverse the 2,414 m buffer zone between research sites; however, six pen-raised quail were trapped on WC after release approximately 1,400 m away from the wild control site.

Nesting/Reproductive Success

Nesting success and predation varied between treatment groups and years, but due to low sample sizes (pen, n = 9; wild/pen, n = 11; wild, n = 11) per year, no conclusive trends can be interfered from these data. Therefore, it is still unclear whether or not pen-raised quail negatively affect native wild quail during breeding attempts and no substantiated data exists on the ecology of offspring from liberated pen-raised and wild quail raised by either a native or pen-raised quail.

Hunting Harvest On Pre-Season Release Sites

Hunting pressure for the three different hunting seasons did not vary greatly by year (541, 461 and 557 hr. hunted); however, the ratio of wild quail harvested per year on the PR site decreased dramatically between each of the three consecutive hunting seasons (1.95, 1.36 and 0.959). The result of lower wild/pen quail harvest rates per hunting season could be due to additive factors such as increased vulnerability to hunter harvest and increases in predation dynamics. However, these data do not justify population estimates or trends due to potential sampling errors such as non-representative harvesting of the "true" quail population that may include hunter avoidance characteristics, habitat selection, flushing and flight patterns, acclimation to feeding sites, loss leg bands, and other factors. Sisson et al. (2000*a*) reported lower survival rates of wild quail on pre-

Table 3: Nesting season data for radio-tagged treatment groups on Groton Plantation, Allendale County, South Carolina for the 1996 and 1997 breeding season.

	Pen-raised		Wild/pen		Wild	
	1996	1997	1996	1997	1996	1997
Hens alive on 6/1	12	8	6	14	7	13
Nests Located	9	9	6	16	10	12
Successful Nests	4	2	4	5	5	2
Avg. Clutch Size	11.4	15.5	11	13	12.5	13
Percent Hatch	97%	97%	94%	94%	96%	81%
Hen to Successful Nest Ratio	33%	25%	67%	36%	71%	15%
Nests Depredated	3	5	1	5	3	5
Nest Abandoned	1	0	1	2	2	3
Nesting Bird Predation	1	2	0	4	0	2

Table 4: Hunter harvest records from Groton Plantation, Allendale County, SC for combined pre-season release sites during three consecutive hunting seasons.

	1996-97	1997-98	1998-99
Hunting Days per Season	133	92	121
Wild Quail Harvested	259	125	116
Pen-raised Quail Harvested	1747	1532	2093
Wild to Pen-raised Quail Ratio	0.191	0.109	0.059
Avg. Wild Quail Harvested[a]	1.95	1.36	0.959
Avg. Pen-raised Quail Harvested[a]	13.14	16.65	17.3

[a] Average Harvest Rate Per Hunting Attempt.

season release sites with hunting pressure; however, again there was no causative link between the release of pen-raised quail and lower survival rates of native quail.

Management Implications

Based on this study, data on survival rates and hunter harvest limitedly indicate that pre-season release programs intended for augmenting hunting populations could potentially affect native quail populations on these sites. Susceptibility to hunting, reduced home range size, changes in habitat selection, and predation factors may be key factors in determining why wild quail populations are decreasing on areas with large-scale pre-season release programs; however, this hypothesis still needs extensive examination before a substantial conclusion can be validated. Linear buffer distances between pre-season release sites and wild control sites may be critical to sustaining large native wild quail populations. Based on our observations, we suggest that a linear buffer distances between 1,400 and 2, 400 m may be required to effectively limit interactions between wild and pen-raised quail.

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Survival and Predation

Survival and Habitat Use of Wild Pheasant Broods on Farmland in Lower Austria

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Wild pheasants (*Phasianus colchicus*) have declined throughout much of their European distribution. The influence of habitat composition on survival and habitat use of wild pheasant broods is not well understood, but is important for population management. We studied the brood-rearing ecology of pheasants using radio-tagged hens on a 2,400 ha farming estate in Lower Austria during 2001-2003. Brood survival , $\geq$1 chick surviving to 3 weeks, (n = 36) were: 74.4% (15.6 SE), 91.9% (7.8 SE), and 65.7% (13.8 SE), during 2001-2003, respectively. Complete brood loss (n = 7) occurred between 2 -17 days after hatching with predation (n = 5) accounting for 71.4% of losses. Survival of broods was influenced by composition of habitats within fixed kernel home ranges . Proportion of planted game crop, mixture of legumes and grasses, within the home range had a positive effect on survival, whereas age and condition of females did not influence brood survival. To improve brood survival rates of pheasants in agricultural landscapes farmers and game managers should consider planting specialist brood rearing mixtures in areas close to nesting habitat.

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Key words: Austria, brood survival, compositional analysis, habitat, home range, *Phasianus colchicus*, wild pheasant

Introduction

Populations of wild pheasant (*Phasianus colchicus*), although widely distributed, are in a state of decline across their European distribution during the past half-century (Hill and Robertson 1988, Tapper 1999, Csànyi 2000) as farming practices have shifted to more intensive row crop agriculture and management targeted at wild pheasants has declined (Jarvis and Simpson 1978, Hill 1985, Potts 1991). Previous research examining population dynamics of pheasants have suggested that at least part of the decline is related to the availability of brood habitat which is important for recruitment (Chiverton 1994, Warner et al. 1999), and brood survival which is a poorly understood component of pheasant life history (Warner et al. 1984, Hill and Robertson 1988).

The most crucial time for broods is the first 14 days post hatch (Hill 1985, Meyers et al. 1988, Riley et al. 1998). Studies suggest that broods select home ranges containing weedy areas and grasslands (Hill 1985), but changes in crop management and pesticide use has reduced weedy plants and insects which are vital for chick development and growth (Potts 1980, Hill 1985, Sotherton et al. 1985, Rands 1985, Sotherton and Robertson 1990). Previous studies of European gamebirds associated with agriculture suggest that abundance of weedy areas, grasslands and insects are inversely related to home range size, and positively correlated with survival of pheasant (Hill 1985), red-legged partridge (*Alectoris rufa*), and gray partridge (*Perdix perdix*) chicks (Green 1984).

Currently, little information on wild pheasant population dynamics is available for Lower Austria, which like the rest of Europe, has seen a precipitous decline in the harvest of wild pheasants during the past 30 years (Draycott et al. 2002). Most research has been undertaken in Britain and North America, but these areas have different farming practices, climate, and other land use composition compared to Austria. Therefore, in this study we examined brood habitat use within home ranges to determine its af-

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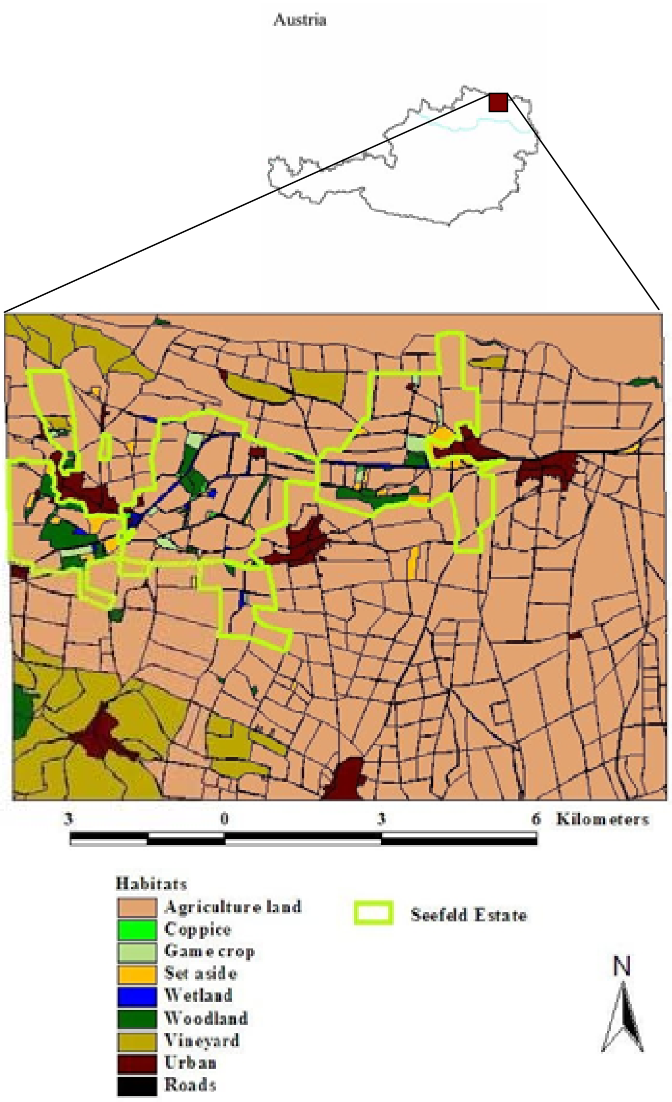

Figure 1: Map of Austria and habitat map of Seefeld Estate and surrounding area, Lower Austria, Austria.

fect upon chick survival within agricultural landscapes of Austria.

Study Area

This study was conducted in the state of Neiderösterich (Lower Austria), Austria on Seefeld estate (1). The estate is a 2,400 ha farm in the town of Seefeld-Kadolz approximately 150 km northeast of Vienna on the border with the Czech Republic. Seefeld estate has been farmed by the Hardegg family since the 15th century and is situated on converted marsh lands with 72% of the estate planted in annual crops. Winter wheat is the dominant crop with an average yield of 5 tons/ha; other crops include barley (winter and summer), sugar beet, potatoes, oil seed rape, and vineyards. Specially planted short-term rotational game crops, long-term set aside (planted grassland), wetland, woodland, and coppice occupy the remaining 28% of the estate. Wine is produced and bottled on the estate along with an indoor pig farm.

The Pulkau River runs the length of the estate and provides water for center pivot irrigation which is distributed by a series of open ditches. During the 1950's the Pulkau was channelized, but since 2000 the meandering flow has been re-established along with wetlands and associated reed (*Phragmites* spp.) beds. Supplemental feed is provided to pheasants by grain hoppers spaced throughout the estate in woodlands and game cover plots during the winter and along woodland edges and hedgerows in the spring to increase the quality of male breeding territories. The red fox (*Vulpes vulpes*), along with crows (*Corvus* sp.) and magpies (*Pica* sp.) are the main predators found on the estate, and are intensively controlled throughout the year.

Seefeld estate has a mid-continental climate with a temperature range of 6 to 37 °C in summer and -25 to 5 °C in winter. Elevation is 190 m and receives an average of 480 mm of precipitation yearly with 160 mm received in May and June. The surrounding land is occupied by small villages, private vineyards and family farms. Family farms have an average size of 14.6 ha and occupy approximately 80% of the land outside the villages (Molterer 1997).

Methods

Pheasant hens were captured from 1 March - 10 April in 2001 - 2003 using baited walk in funnel traps. Captured hens were aged, weighed, tarsus measured, and fitted with a numbered aluminum patagial tag, and a 9.9 g necklace collar (Holohil model RI-2B). The condition of each hen was determined by the condition index established by Robertson et al. (1985). Radio-tagged hens were located 3 times weekly by radio telemetry until nesting at which time they were located every other day. Once a nest hatched it was examined to determine number of chicks that hatched. Broods were located twice daily from a distance of $\geq$15-30 m for the first 21 days to determine exact habitat use. A brood was considered lost if a brood caution or gathering call (Giudice and Ratti 2001) were not heard during consecutive observations or if the hen died.

We calculated a UTM coordinate from topographic maps for each brood location which was overlaid on a habitat map using ArcView 3.1. Bootstrapping with replacement was performed using the Animal Movement Extension 2.04 (Hooge and Eichenlaub 2000) to estimate number of locations needed to construct home ranges. The 100% minimum convex polygon (MCP) (Mohr 1947) and 95% fixed kernel (Worton 1989) home range were calculated using the Animal Movement Extension 2.04 (Hooge and Eichenlaub 2000) for the first 21 days post hatch.

We used analysis of variance (ANOVA) for unequal sample size (Sokal and Rohlf 1969) using PROC GLM (SAS Institute, Inc. 1999) to determine if home ranges differed between year and age, Tukey's test was used to compare post-hoc results.

Proportion of habitats within each home range was calculated in ArcView and compositional analysis (Aebischer et al. 1993) was used to estimate habitat preference at the 2nd and 3rd order (Johnson 1980) using BYCOMP.SAS (Ott and Hoovey 1997) for MCP and 95% fixed kernel home ranges. Wilk's λ was used to determine if habitat use was not random by running 1000 iterations of the data; habitat preference was ranked by a series of paired *t*-tests.

For home range analysis, land cover on the estate was combined into 4 categories to represent land use patterns that should have biological significance to pheasants (Aebischer et al. 1993); 1) Agriculture including all row crops and vineyard (89%), 2) set aside: planted grassland and game crop (1.9%), 3) wetland shrub (1.2%), and 4) woodland: wooded areas, coppice, and wind breaks (2.1%). Any values missing at the 2nd or 3rd order were replaced following criteria established by Aebischer et al. (1993).

Brood survival (the proportion of broods in which at least one chick survived to fledging) was calculated using the Kaplan-Meier method ((Kaplan and Meier 1958) using the known fate model in Program Mark (White and Burnham 1999) using the logit scale for each year 2001-2003. Broods were left censored and constant survival (S[.]) model estimate was used.

Table 1: Habitat ranking matrix of 4 defined habitat types based upon 2nd order (A) and 3rd order (B) compositional analysis of MCP home ranges. Higher ranking indicates greater use compared to availability. Within the matrix, a (+) means that the row habitat is used relatively more than the column habitat, whereas a (-) means the opposite and a +++ or — mean that they are different at (P<0.05).

A.

Habitat	**Woodland**	**Set aside**	**Wetland**	**Agriculture**	**Rank**
Woodland	.	—	—	—	0
Set aside	+++	.	+	-	2
Wetland	+++	-	.	-	1
Agriculture	+++	+	+	.	3

B.

Habitat	**Woodland**	**Set aside**	**Wetland**	**Agriculture**	**Rank**
Woodland	.	—	+	+	2
Set aside	+++	.	+++	+++	3
Wetland	-	—	.	+	1
Agriculture	-	—	-	.	0

Non-habitat variables and landscape variables were then used as covariates within the model to determine their affect upon brood survival. Non-habitat variable include age and condition of hen at time of capture. Landscape variables include nest habitat and proportion of agricultural land, game crop, set aside, wetland, woodland, and amount of edge (m/ha) within each home range. Habitat proportions calculated within 95% fixed kernel ranges were used since a minimum of 10 locations can be used (Kenward 2001). Broods with less than 10 locations the arithmetic mean was calculated then buffered by the average 95% fixed kernel home range. Edge was calculated using Patch Analyst 3.1 (Rempel and Carr 2003). Survival constant (S[.]) and by year (S[g]) were chosen *a priori* to determine the effect the covariates had upon survival since covariates were not measured over time and broods were left censored. To determine which models fit best and the effect of each covariate upon survival Akaike's Information Criteria for small sample size (AIC_c) was used (Anderson et al. 2000). Slope (β), unconditional standard error (SE) and 95% confidence interval (CI) were calculated by model averaging for each covariate. If the CI for a covariate included zero we considered it to have no influence on survival.

Results

One hundred and twenty-seven pheasant hens were radio-tagged during 2001-2003 which produced 36 broods. Home range size for successful broods (n = 28) was 11.1 ha (±2.13 SE) and 14.6 ha (±2.45 SE) for MCP and adaptive kernel methods, respectively. No difference was found in home range between years ($F_{2,25}$ = 1.99, P = 0.16), or age of the hen ($F_{1,26}$ = 0.02, P = 0.90).

Our habitat analysis suggested that habitat use by hen pheasants with broods was not random at the 2nd (Wilk's λ = 0.59, $F_{3,30}$ = 6.92, P = 0.001) or 3rd (Wilk's λ = 0.44, $F_{3,30}$ = 12.66, P < 0.0001) order for MCP home ranges. At the 2nd order agricultural land ranked highest, but 3rd order analy-

Table 2: Habitat ranking matrix of 4 defined habitat types based upon 2nd order (A) and 3rd order (B) compositional analysis of 95% fixed kernel home ranges. Higher ranking indicates greater use compared to availability. Within the matrix, a (+) means that the row habitat is used relatively more than the column habitat, whereas a (-) means the opposite and a +++ or — mean that they are different at (P ¡ 0.05).

A.

Habitat	**Woodland**	**Set aside**	**Wetland**	**Agriculture**	**Rank**
Woodland	.	—	—	—	0
Set aside	+++	.	+	-	2
Wetland	+++	-	.	-	1
Agriculture	+++	+	+	.	3

B.

Habitat	**Woodland**	**Set aside**	**Wetland**	**Agriculture**	**Rank**
Woodland	.	-	-	+++	1
Set aside	+	.	-	+++	2
Wetland	+	+	.	+++	3
Agriculture	—	—	—	.	0

sis showed that set aside was most preferred habitat within home ranges (1). Adaptive kernel home range results indicated that habitat use was not random at the 2nd (Wilk's λ = 0.66, $F_{3,30}$ = 5.20, P = 0.005) or 3rd (Wilk's λ = 0.64, $F_{3,30}$ = 5.69, P = 0.003) order. Habitat rankings at the 2nd order were identical to MCP with agricultural land ranked highest, but 3rd order differed from results for MCP in that wetland habitat was preferred (2).

Survival of broods for the first 21 days was estimated at 74.4% (±15.6 SE), 91.9% (±7.8 SE), and 65.7% (±13.8 SE) in 2001, 2002, and 2003, respectively (2). Over the 3 years 7 complete broods were lost between 2-17 days after hatching with average loss occurring 11 days after hatch (±2 days SE). Predation by fox (n = 4) and other mammalian predation (n = 1) were responsible for the loss of 5 (71.4%) broods; the other 2 (28.6%) were lost to exposure and during harvest operations.

Non-habitat covariates estimated did not have an affect upon brood survival. Game crop was the only habitat variable with a 95% CI (β = 609.04, 95% CI 472.3 - 745.8) that did not include zero. Our model showed broods that utilized game crop had a 100% survival. For all other habitat covariates tested, the 95% CI contained zero, but several were highly skewed. Although not significant woodland and long-term grassland set-aside appeared to have a negative impact upon brood survival, whereas our data suggested wetlands had a positive relationship to brood survival (3).

Discussion

We found that pheasant broods in agricultural landscapes of Austria had home ranges greater than the 4.8 ha (Hill 1985) reported for England and were at the upper end of the range of 2 - 11 ha observed in the U.S. (Kuck et al. 1970, Hanson and Progulske 1973, Warner 1979). One probable reason broods had larger home ranges than observed in England is that broods were followed for 21 instead of 14 days. Hanson and Progulske (1973) found that home range size increases with brood age.

Habitat selection at the 2nd order showed that agricultural land was incorporated into home

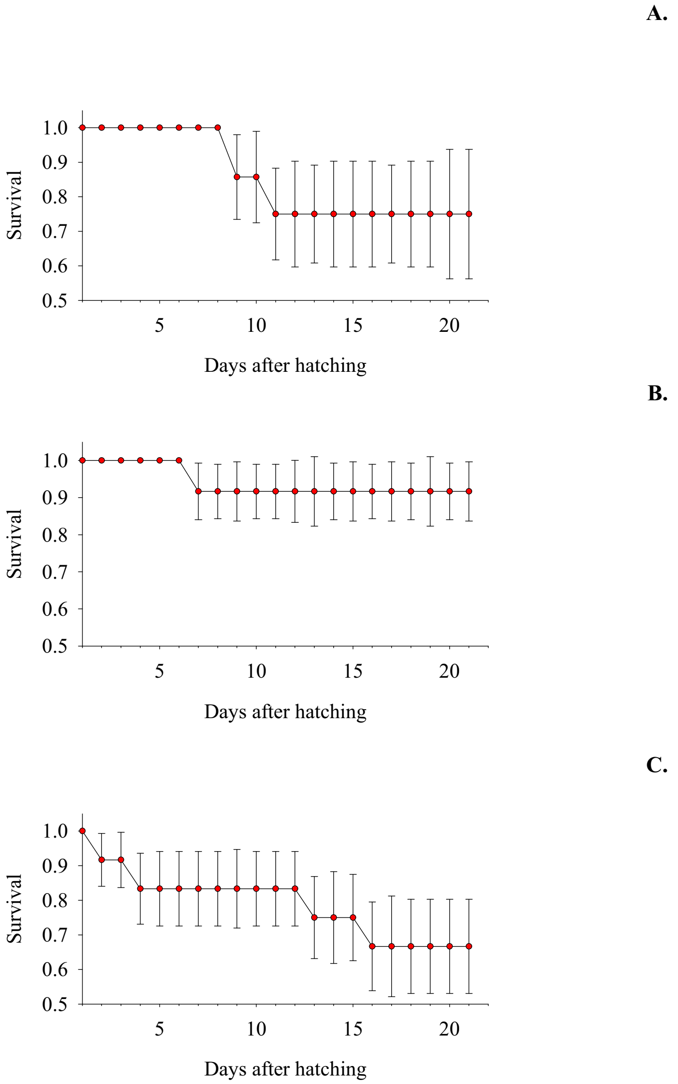

Figure 2: Kaplan-Meier survival estimates (+ SE) of radio-tagged pheasant broods during (A) 2001 (n = 9), (B) 2002 (n = 14), and (C) 2003 (n = 12) at Seefeld Estate, Lower Austria, Austria.

ranges and supports previous research that observed pheasant broods in cereal crops (Warner 1979, Hill 1985, Enck 1986). Aebischer and Blake (1994) suggested that the structure of cereal crops allows for easy movement of broods and provides protection from predators and that the first 5 m of the field edge also contain more insects and weeds (Chiverton 1994) where gray partridge broods are often found (Green 1984). Third order habitat analysis revealed that set aside was preferred habitat. Aebischer and Blake (1994) reported that properly managed set aside has a greater diversity of plants that attract a wide range of insects and produce small seed that is critical for chick development. This corroborates other studies that found broods in undisturbed grassland and weedy areas (Warner 1979, Hill 1985, Riley et al. 1998).

We found that brood loss occurred during the

Table 3: AICc value, delta AICc, slope (β), and 95% Confidence Interval (CI) of non-habitat and habitat covariates upon 21 day survival of radio-tagged pheasant broods at Seefeld Estate, Lower Austria, Austria during 2001 - 2003. Inclusion of zero within the 95% CI suggests there is no significant slope.

				95% CI	
Pheasant hen covariates	AICc	Δ AICc	Slope (β)	Lower	Upper
Condition index	80.247	0	0.187	-0.308	0.683
Age	80.289	0.042	0.382	-1.222	1.986
Habitat covariates					
Game crop (%)	74.516	0	609.037	472.284	745.79
Wetland (%)	75.735	1.219	36.578	-9.696	82.851
Edge (m/ha)	75.995	1.479	-0.347	-0.929	0.235
Woodland (%)	78.777	4.261	-14.586	-32.924	3.753
Agriculture (%)	80.558	6.042	-1.128	5.265	3.009
Set aside (%)	80.801	6.285	-0.219	-3.604	3.167

first 17 days post hatch and that most losses were attributable to mammalian predation, with minor loss to exposure and harvest. This is similar to previous studies that found brood loss was greatest within 14 days after hatching (Hill 1985, Meyers et al. 1988, Riley et al. 1998). Mammalian predation has been implicated as an important cause of brood loss in previous studies (Riley et al. 1994, 1998). Other studies in Iowa found that the dominant predator upon pheasants is the red fox (Riley and Schulz 2001). Losses to avian predation has been observed in other studies (Carroll and Sayler 1990), but was not observed during our study. Removal of predators can increase recruitment while implemented, but often return to pre-treatment levels when predator removal ceased (Chesness et al. 1968, Jensen 1970).

Brood survival rates of 65-92% we observed during the first 21 days are similar to survival rates reported in North America (Gates and Hale 1974, Warner et al. 1984, Carroll and Sayler 1990, Riley et al. 1998, Nohrenburg 1999), but greater than values reported in the United Kingdom (Hill 1985). In Illinois and Iowa abundance of grassland was correlated to increased chick survival (Warner et al. 1984, Riley et al. 1998). Game crop likely offers concealment from predators and a higher abundance of arthropods over cereal crops (Sotherton et al. 1985). At Seefeld Estate the planted game crops contained a number of different species including legumes and grasses planted at low seeding rates and were managed to provide both an abundance of food and the correct structure to allow ease of movement of broods through the base of the crop. Conversely, the permanent grassland set-aside areas were based on tussock forming grasses including cocksfoot (orchardgrass *Dactylis glomerata*) which although ideal for nesting (Bliss 2004) are not suitable for young foraging broods. Indeed our data suggested that there may be a negative relationship between survival and permanent grassland set-aside. Our results also suggest that wetland habitat may positively affect survival, since it may provide cover and food for broods once crops are harvested. Woodland seemed to negatively affect brood survival and support results from Hill (1985). Woodland edge can negatively impact herbaceous vegetation in adjacent habitats and increase the number of predators (Wasilewski 1986) and has been shown to affect habitat selection by pheasants (Wasilewski 1986) and gray partridge (Dudzinski 1992).

Management Implications

We advocate habitat management in conjunction with predator control to enhance pheasant populations on in Lower Austria. During brood rearing season hens with broods preferred set-aside habitat within their home ranges and those that utilized game crop had 100% survival. When game crop is properly managed it has low stem density and contains little ground debris which allows for easier movement. Game crop also attracts a wider variety of insects than found in crops which is important for chick survival (Hill 1985). We also determined that woodland habitat had a detrimental affect upon survival and therefore set aside and game crop habitat should not be placed next to woodland.

We found that the majority of brood loss was to mammalian predation even though predation control is exercised on Seefeld Estate. Therefore we advocate further research on broods by marking individual chicks at time of hatch as (Riley et al. 1998) conducted in Iowa. This would allow for a detailed estimate of chick survival and impact of predators upon broods to be assessed.

This work in conjunction with habitat management for winter, breeding, and nesting and future chick research at Seefeld Estate will allow for development of management plans for wild pheasant populations in their mid-European distribution.

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A Review of Crippling Loss for Northern Bobwhites

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Many studies have reported estimates of crippling loss (i.e., birds shot, noticeably or not, and not retrieved) for northern bobwhites (*Colinus virginianus*); however, comparisons among studies have been difficult because of a lack of standardized definitions and methods of calculation. The purpose of this paper was to: 1) provide a review of crippling loss of bobwhites across their geographic range, and 2) develop terminology that allowed for explicit discussion of crippling loss and facilitated comparison among studies. We also obtained an estimate of crippling loss for bobwhites in southern Texas using data from a larger study investigating the effects of ranch-road baiting on bobwhites. Reported estimates of crippling loss ranged from 5 to 31% of recorded harvest and 5-24% of total kill. We propose that studies reporting crippling loss use explicit definitions including those developed herein, allowing for inter-study comparisons. Documenting crippling loss in the field should include monitoring of radio-marked bobwhites the morning after a hunt to correctly identify crippled loss birds. In addition, practices (e.g., amount of time spent looking for downed birds) potentially minimizing crippling loss on harvested bobwhite populations should be identified.

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Key words: crippling loss, harvest, northern bobwhite, population,Texas

Introduction

Despite the occurrence of crippling loss during harvest, many studies discussing bobwhite harvest mortality have failed to mention crippling loss in their estimates of harvest rates (Vance and Ellis 1972, Hurst and Warren 1982) or cause-specific mortality (Curtis et al. 1989, Robinette and Doerr 1993, Dixon et al. 1996, Williams et al. 2000, Madison et al. 2002). These studies either did not consider crippling loss significant to their analysis, had no crippling loss to report, or included crippling loss within their estimate of harvest rate but did not report it. If crippling loss was included in these estimates but not mentioned, no specification was given as to what percentage of harvested bobwhites represented birds lost to crippling.

The purpose of this paper was to 1) provide a review and comparison of crippling loss of bobwhites across their geographic range using standardized terminology, and 2) develop terminology that allows for explicit discussion of crippling loss and facilitates comparisons among studies. We also provide an estimate of crippling loss for bobwhites in southern Texas obtained from data gathered as part of a larger study investigating the effects of ranch-road baiting on bobwhites (Haines et al. 2004).

Methods

We conducted a literature review of bobwhite studies which specifically reported crippling loss as a cause-specific mortality. We summarized crippling loss for bobwhites in tabular form by region.

Crippling Loss Estimate

We obtained an estimate of crippling loss for southern Texas on a private ranch located 8 km east of Hebbronville, Texas in Jim Hogg County. The study area is contained within the Rio Grande Plains ecoregion (Gould 1975). Annual rainfall ranges from 35 to 66 cm and soils range from clays to sandy loams (Correll and Johnston 1979). Haines et al. (2004) provided a more complete description of the study area.

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We trapped bobwhites from mid-August through September 2001 and 2002 using funnel traps baited with milo (Stoddard 1931, pp. 442-445) and by night-netting roosting coveys (Labisky 1968) during November through January 2001-2002 and 2002-2003. We banded all captured bobwhites and radiocollared any bobwhite weighing ≥150 g. We fitted bobwhites with 6 to 7g neck-loop radio-transmitters (American Wildlife Enterprises, Tallahassee, Florida). We monitored each radiomarked bobwhite 5 times a week to determine cause-specific mortality. Bobwhites were found by homing (White and Garrott 1990, Stauffer 1993). We categorized bobwhite mortality as predation or harvest following the criteria of Carter et al. (2002). We defined crippling loss as any radio-marked bobwhite found dead and intact (i.e., not depredated or scavenged) following a hunt. We monitored radiomarked bobwhites within 2-48 hours following a hunt. We confirmed crippling loss by removing the feathers from the body of the recovered bobwhites and documenting the presence of shotgun pellet wounds.

Results

Literature Review

The interpretation of what constitutes crippling loss, and therefore its calculation varied across studies (Table 1). Bennit (1945) defined crippling loss as the number of birds that were shot but unretrieved from the field, whereas Kellogg and Doster (1971) not only included the number of birds that were shot and unretrieved, but also the number of birds that were "feathered" by shot but continued to fly. Parry et al. (1997) defined crippling loss as the number of birds found in the field dead from pellet wounds (via radiotelemetry) post-hunt. The calculation of crippling loss has also varied among studies, being calculated either as a proportion of retrieved birds (e.g., Bennit 1945, Suchy and Munkel 2000) or as a proportion of total kill (retrieved + unretrieved; e.g. Parry et al. 1997, Lehmann 1984).

Crippling Loss Estimate

We captured and radiomarked 150 bobwhites from mid-August to January 2001-2002 and 2002-2003. We documented 53 mortalities of radiomarked birds, of which 34 (65.4%) were depredated, 18 (32.7%) were harvested, and 1 (1.9%) died of unknown causes. Of the 18 harvested bobwhites, 12 were retrieved and 6 were unretrieved (found via radiotelemetry). Thus, crippling loss represented 50% (6/12) of retrieved harvest and 33% (6/18) of total kill. Of the 6 crippled losses, 1 represented a bobwhite which was noticeably shot but survived. This radio-marked bobwhite was shot through the wing at the base of the primary feathers and lost the ability to fly. It survived 7-9 days in the field after being shot until it was eventually depredated. Exclusive of our crippling loss estimates, reported estimates for crippling loss in the literature ranged from 5-31% of retrieved harvest and from 5-24% of total kill (Table 1).

Discussion

Definitions

When defining crippling loss we are interested on how it impacts the total mortality rate due to harvest in the absence of natural mortality. Thus, we suggest representing all hunting-related mortality occurring from start to end of a hunting period, including both retrieved and unretrieved animals, as H_o. We express H_o as

$$H_o = H_r + H_l + H_f + H_w \qquad \text{(Equation 1)}$$

where

H_r = mortality rate of harvested animals noticeably shot, downed, and retrieved (i.e., "bagged"; retrieval rates),

H_l = mortality rate of harvested animals noticeably shot, downed, but not retrieved (i.e., lost),

H_f = mortality rate of animals noticeably shot but not downed (i.e., "feathered"), not retrieved, and subsequently dead as a result of shot wounds,

H_w = mortality rate of animals not noticeably shot, not downed, not retrieved, and subsequently died as a result of shot wounds.

Three of the variables (H_r, H_l, and H_f) can be read-

Table 1: Summary of published studies reporting percentage of crippling loss for northern bobwhite.

Studies by region	Location of study	Retrieved harvest	Reported crippled	Total harvest	Crippling loss (%) of retrieved harvest	Crippling loss (%) of total harvest
Southeast						
Gehrken (1956:241)	Southeast Virginia	NR[c]	NR	NR	10.7[d]	NA[f]
Rosene (1969:185)	South Carolina	500	100	600	20	16.7
Kellog and Doster (1971:148)[a]	Florida	846	266	1,112	31.4	23.9
Doster et al. (1982:47)	Florida	2,245	615	2,860	27.4	21.5
Lehmann (1984:304)	Georgia	8,187	431	8,618	5.3	5
Mid-west						
Bennitt (1945)	Missouri (Metropolitan hunters)	NR	NR	NR	8.7[e]	NA
Bennitt (1945)	Missouri (Other hunters)	NR	NR	NR	7.4[e]	NA
Hood (1955:16)[a]	Mississippi	535	135	670	25.2	20.1
Roseberry and Klimstra (1984:41)	Illinois	2,893	349	3,242	12.1	10.8
Suchy and Munkel (2000)	South-central Iowa (BSSA)	NR	NR	NR	28	NA
Suchy and Munkel (2000)	South-central Iowa (MSA)	NR	NR	NR	17	NA
Burger et al. (1995:405)	Missouri	121	28	149	23.1	18.8
Southwest						
Parmalee (1953:343)	East-central Texas	243	64	307	26.3	20.8
Snyder (1978:34)	Colorado	1,535	242	1,755	16	13.8
Shupe et al. (1990:24)	South Texas	NR	NR	NR	NA	15.1
Parry et al. (1997)	W. Oklahoma (hunter reported)	5,108	805	5,913	15.8	13.6
Parry et al. (1997)	W. Oklahoma (radiomarked)	150	20	170	13.3	11.8
Haines (2004)[b]	S. Texas	12	6	18	50	33.3

[a] crippled estimate includes both 1) bobwhites downed but not retrieved and 2) feathered and not retrieved; Kellog and Doster (1971; 109 + 157, respectively) and Hood (1955; 57 + 87, respectively).

[b] crippled estimate includes 5 bobwhites downed but not retrieved and 1 bobwhite noticeably shot but dying 7-9 days later presumably from predation.

[d] referenced as % shot and lost; presumed of harvested birds and not total kill.

[e] referenced as % of bag shot down but lost; presumed of harvested birds and not total kill.

[f] not applicable.

ily estimated from harvest data, provided these data are recorded in the field. The fourth variable, H_w, is more difficult to measure but can be estimated with the aid of radiotelemetry. It is important to note that the latter 3 parameters of Equation (1) constitute all the variables contributing to crippling loss; thus crippling loss rate can be expressed as

$\mathrm{C} = H_l + H_f + H_w$ (Equation 2),

and harvest rate (H_o) in Equation (1) can be simplified to

$H_o = H_r + \mathrm{C}$ (Equation 3).

Crippling loss proportions can be calculated as a proportion of retrieved birds, as a proportion of total kill, or as a proportion of the prehunt population (N_f). These estimates are calculated as

$\mathrm{C}_r = \mathrm{C} / H_r$ (Equation 4),

$\mathrm{C}_k = \mathrm{C} / H_o$ (Equation 5),

and

$\mathrm{C}_n = \mathrm{C} / N_f$ (Equation 6),

respectively. Only the former 2 definitions of crippling loss (i.e., C_r and C_k) have been reported in the literature.

Crippling Loss Estimate

Our field estimates of crippling loss, whether in reference to retrieved harvest or total kill, were higher than estimates reported in the literature (Table 1). Our higher estimates of crippling loss may be consequences of inadequate sample size, low harvest rate, and/or our radiomonitoring of bobwhites 2-48 hours following hunts. The rather "immediate" monitoring of radio-marked bobwhites after a hunt might have provided us with a more accurate depiction of actual crippling loss, as bobwhites that would have been misclassified as "depredated" in a less timely schedule would be correctly classified as "crippled loss". However, because monitoring took place 2-48 post hunt our estimates of crippling loss could still potentially be low, with birds being lost to scavengers. In addition, Parry et al. (1997) located radio-marked bobwhites the morning after a hunt and documented no difference between hunter-reported and radiotelemetry-generated estimates of crippling loss (Table 1). However, we could not compare between hunter-reported and radiotelemetry-generated estimates of crippling loss because the landowner of our study site did not keep records of unretrieved or feathered birds.

The protocol of Parry et al. (1997) to monitor bobwhites the day after a hunt represents a good approach to document crippling loss in harvested populations. However, we propose that monitoring of radio-marked bobwhites should continue on a daily basis beyond 1-day post hunt because birds that were crippled but survived $>$1 day may be classified incorrectly, as these handicapped birds would not be found dead until they succumbed to injuries or were subsequently depredated. Bobwhites that were shot but only crippled are more susceptible to predation (Curtis et al. 1989, Suchy and Munkel 2000). We documented this for the bobwhite that was shot through the wing but still managed to survive an additional 7-9 days after the hunt. In addition, we recommend more research to validate the findings of Parry et al. (1997) that there was no difference between hunter-reported and radiotelemetry-generated estimates of crippling loss.

Suggestions

We propose that studies reporting crippling loss use terminology which explicitly specifies the type of crippling loss being calculated (i.e., proportion crippling loss of retrieved harvest, of total kill, or of prehunt population) as well as report its composition. This information then could be used to more accurately compare among studies and allow for better evaluation of the effects of harvest on bobwhite populations (Pollock et al. 1989, Roseberry 1979, Roseberry and Klimstra 1984).

Crippling loss could substantially change the estimation of harvest impacts on wild bobwhite populations, especially if harvest rates did not account for this added mortality. Practices that are most effective at mitigating crippling loss (e.g., skill of hunter, dog training, number of hunters in a party, amount of time spent looking for downed birds, tighter grouping of pellet shot, etc.) should be identified. In addition, future research should be conducted to model the impacts of crippling loss on harvested bobwhite population dynamics, to measure the impacts of hunting with and without measured crippling loss, and to measure the impacts of crippling loss at varying levels of harvest effort.

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Fate and Survival of Radio-marked Montezuma Quail

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Montezuma quail (*Cyrtonyx montezumae*) represent one of the least studied North American quail species, particularly in Texas. This lack of information may be partly due to their secretive nature and difficulty of capturing. We provide the first published report of fate of radio-marked Montezuma quail in Texas. We captured, radio-marked, and released 14 Montezuma quail on Elephant Mountain ($n = 9$) and Davis Mountain Preserve ($n = 5$) during 2000-2005. We used 2 methods of attachment for pendant style neck-loop radio transmitters. Body-loop transmitters were affixed to quail captured at the Davis Mountain Preserve whereas both body-loop and neck-loop were used at Elephant Mountain. All radio-marked Montezuma quail died within a relatively short period (1-16 days). Causes of mortality for most Montezuma quail were attributed to raptors ($n = 9$), mammals ($n = 1$), and miscellaneous ($n = 4$). Because this low survival rate would not sustain a natural population, we suspect trapping, handling, and/or radio-marking negatively affected survival. It is possible that transmitters potentially restricted escape movements or interfered with other behavior thereby making Montezuma quail more vulnerable to predation. Traditional techniques used to affix radio transmitters or transmitter design itself need to be refined if Montezuma quail are to be studied using radio telemetry.

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Key words: *Cyrtonyx montezumae*, Montezuma quail, radio-marked, radio telemetry, survival

Introduction

Montezuma quail (*Cyrtonyx montezumae*) are secretive birds that are associated with pine-oak woodlands of the desert southwest (Stromberg 2000). In New Mexico (Holdermann 1992) and Arizona (Heffelfinger and Olding 2000), Montezuma quail are classified as game birds and provide a valuable source of recreational opportunities for hunters. In Texas, Montezuma quail are also classified as game birds but have a closed season. Throughout the United States, Montezuma quail are appreciated and sought by bird watchers and other outdoor enthusiasts. Contrary to their monetary or intrinsic value, little information exists on the ecology of Montezuma quail. Most ecological information on Montezuma quail has been inferred from reports from early naturalists (Fuertes 1903, Ligon 1927, Leopold and McCabe 1957) or from harvested samples (Bishop and Hungerford 1965, Brown and Gutiérrez 1980). Only one study (Stromberg 1990) has provided data on the population ecology of Montezuma quail using contemporary techniques (e.g., radiotelemetry).

The advent of radiotelemetry has had a profound effect on the ability of wildlife biologists to obtain valuable information on the ecology of various wildlife species (Samuel and Fuller 1996). In fact, radiotelemetry has allowed researchers to refine the life history of several quail species. For example, the reproductive strategy for northern bobwhite quail (*Colinus virginianus*) was labeled as "monogamous" by many early researchers (Stoddard 1931, Klimstra and Roseberry 1975). It was not until extensive research using radiotelemetry that biologists discovered that northern bobwhite had a very complicated reproductive strategy that implements a combination of monogamy, polyandry, double clutches, and multiple brooding (Curtis et al. 1993, Burger et al. 1995).

One of the major assumptions of using ra-

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diotelemetry is that the transmitter does not affect the animal's movement or survival. Collectively, manufacturers and biologists have developed light weight transmitters that are affixed with minimal effect on the behavior and survival of the animals. For quail, transmitters are designed to be $<5\%$ of the quail's body mass and are typically affixed as neckloops or backpacks (B. Mueller, American Wildlife Enterprises, Inc., Tallahassee Florida, personal communication). Stromberg (1990) provides the only published account of using radiotelemetry on Montezuma quail. Stromberg (1990) postulated that because of the unique foraging style of Montezuma quail (e.g., digging), the poncho-style transmitter interfered with their behavior and thus survival.

In 2000 we initiated a study investigating the ecology of Montezuma quail in the Trans-Pecos ecoregion of Texas. Because little information exists on the use of radiotelemetry on Montezuma quail, we provide a detailed account on the handling, radiotagging, transplanting, and subsequent fate of radio-marked Montezuma quail.

Study Area

This study was conducted on Elephant Mountain Wildlife Management Area (WMA) and the Davis Mountain Preserve. The two study sites are approximately 108 km apart and located in the Trans-Pecos ecoregion of Texas. Precipitation for both sites varied form 38-51 cm at Elephant Mountain to 45-58 cm at the Davis Mountain Preserve. Most of the precipitation fell as torrential rains during the months of July-August. Other precipitation occurred in the form of snow during December and January.

Elephant Mountain WMA is located 40 km south of Alpine, Texas in Brewster County. The actual study location was the summit and slopes of Elephant Mountain which is situated in the center of the WMA. Elephant Mountain ranges in elevation from 1,615 m in the north side to 1,900 m in the southern end and rises approximately 609 m above the surrounding table lands. Vegetation found at Elephant Mountain WMA was typical of the Chihuahuan Desert. The lowlands surrounding Elephant Mountain were composed of desert scrub species and the top of Elephant Mountain was desert grassland interspersed with small shrubs. The small shrubs include oak (*Quercus* spp.), mountain laurel (*Sophora secundiflora*), and fragrant sumac (*Rhus aromatica*).

The Davis Mountain Preserve encompasses 7,300 ha of West Texas montane habitat in Jeff Davis County approximately 25 km northwest of Fort Davis, Texas. Elevation ranges from 1,700 m to over 2,000 m. The lower elevations were composed of mainly oak savannah vegetation. The dominant grasses included cane bluestem (*Bothriochloa barbinodis*), side-oats grama (*Bouteloua curtipendula*), and blue grama (*Bouteloua gracilis*). Typical woody species included ponderosa pine (*Pinus ponderosa*), white pine (*Pinus strobiformis*), Mexican pinyon (*Pinus cembroides*), and alligator juniper (*Juniperus deppeana*).

Methods

A total of 14 Montezuma quail were captured, radio-marked and released on Elephant Mountain ($n = 9$) and Davis Mountain Preserve ($n = 5$) during 2000 and 2005, respectively. The methods employed to capture quail included mist nets, trained bird dogs and fishing dip nets, and night-netting.

The mist net capture method consisted of visually detecting quail, carefully approaching and slowly lowering the mist net over the quail. The observers would then quickly remove the captured quail from the mist net and proceed to data collection (e.g. sex, age, weight, etc.)

Trained bird dogs were used to aide in detecting quail. Once the dog located the quail (on point), 2-3 observers with fishing dip nets would approach the point location. To capture the quail, the observers would either visually detect the quail and lower the dip net, or capture on the wing as they flushed up.

Night netting involved having at least one radio-marked quail in the covey. The radio-marked quail were located (via radio telemetry) at night while on roost and a general location (2m x 2m area) was determined by triangulation. The exact location was revealed by shining a spotlight on the general location until the roosting covey was detected. A fishing net was then slowly lowered on the covey.

All quail captured were aged, sexed, banded, measured, and fitted with a radiotransmitter at time of capture. Quail that weighed >150 g were fitted with either a neck-loop or body-loop style transmitter (American Wildlife Enterprises, Tallahassee, Florida). Handling time was maintained to <5 minutes to minimize stress. Quail that were captured and to be transplanted were kept in a cardboard holding box (46 x 46 x 76 cm) containing grain sorghum and water and transported to a lab. The quail remained undisturbed in captivity overnight and were subsequently transported to release sites. Transplants were conducted to supplement trapping efforts at both study sites. We had experienced minimal to no trapping success at Elephant Mountain as well as at the Davis Mountain Preserve.

At the Elephant Mountain release sites, the quail were released as pairs. Montezuma quail within each pair were separated approximately 100 m to encourage calling between both birds (i.e. assembly call) in hope of attracting surrounding quail. The reasoning was that they would assemble with resident quail. Montezuma quail released at the Davis Mountain Preserve were released as single birds and relocated 300-750 m from original capture site. For transplanted quail, total time elapsed from time of first capture to time of release was <24 hours. For discussion purposes, captured Montezuma quail are referred to by band number (e.g., MQ140) throughout this manuscript.

Results

Elephant Mountain WMA

An adult male Montezuma quail (MQ138) was captured at Elephant Mountain on 20 December 2000 with the aide of a bird dog and a fishing dip net. It was banded and fitted with a neck-loop transmitter. Attempts to radiolocate MQ138 the following day and subsequent ground and aerial searches failed. Approximately 6 months later, the transmitter was located in a large tree along a bluff. Based on the location of the transmitter and the "pig-tailed" appearance of the antenna, it was concluded that MQ138 was preyed upon by a raptor.

Five Montezuma quail (MQ140, MQ141, MQ142, MQ143, and MQ144) ventured onto Sul Ross State University campus on 22 January 2001 and were captured using portable mist nets. Also, taking advantage of the quail's primary defense strategy of laying motionless and hunkering down, some birds were captured by hand. The quail were placed in a cardboard holding box where they remained undisturbed and in captivity overnight.

The following day an adult female (MQ141) was fitted with a body-loop transmitter and transported to the original capture site and released where 3 covey mates avoided capture the day prior. A day later, the carcass of MQ141 was found intact <100 m from release site. Her mortality is attributed to handling stress or exposure.

The remaining 4 quail were translocated to Elephant Mountain, radio-tagged, and released at 2 sites where fresh quail sign (e.g., diggings) had been located. They were released as pairs (F, F and M, F). The two females (MQ144, MQ140) were radio-tagged with a neck-loop (MQ140) and chest-loop (MQ144) style transmitters and released. They were radiolocated and found dead <150 m from release site and <40 m apart from each other the day following the release. The "pig-tailed" antenna indicated raptor predation.

The male-female pair (MQ143, MQ142) was radio-tagged with body-loop style transmitters, released, and located the following day with 3 resident Montezuma quail. MQ143 and MQ142 were recaptured along with the 3 resident quail using the night-netting technique. Because we had been successful at integrating MQ143 and MQ142 with resident quail, we attempted the integration process again by relocating the pair to another release site. MQ143 and MQ142 were found dead the day following their second release. The cause of mortality for both quail was attributed to raptors.

The 3 resident Montezuma quail (MQ145, MQ146, and MQ147) were all fitted with neck-loop style transmitters and released at their original capture site. They were found dead the day after their release. Clipped body parts (i.e., leg, wing, meat stripped off of bones) suggested avian kill.

Davis Mountain Preserve

A female (MQ101) was flushed and caught using a handheld fishing dip net on 25 June 2005. MQ101 was morphologically measured, radio-marked, and released on site. MQ101 subsequently paired up with 3 different males on three separate occasions. These 3 males (MQ104, MQ102, and MQ103) were all captured using the night-netting technique. All quail were kept in captivity in a cardboard holding box overnight and released the morning following their capture. Given the time of year (Montezuma quail pairing season) all quail captured with MQ101 were relocated from release site to prevent pairing up and to encourage MQ101 to pair up with a different male Subsequently, this would allow us to night-net other quail and increase our sample size.

MQ101 survived for 15 days and its mortality was attributed to canid predation. MQ104 was captured with MQ101 on 27 June 2005 and released as a single 750 m from capture site the following morning. MQ104 never joined other Montezuma quail and survived for 12 days. This mortality is attributed to avian predation.

MQ102 was captured with MQ101 on 28 June 2005 and released as a single 700 m from capture site the following morning. MQ102 never joined other quail and survived for at least 12 days. The transmitter signal was lost thereafter and assumed that the signal loss was due to transmitter failure.

MQ103 was captured with MQ101 on 2 July 2005 and released as a single 300 m from capture site the following morning. MQ103 was radiolocated for 16 days until the carcass was found with 1 leg missing.

A fifth Montezuma quail (MQ105) was captured using a trained dog and handheld fishing dip net on 28 October 2005. MQ105 was radiolocated for 3 days and then signal was lost. The transmitter was located 18 days later. The cause of mortality for MQ105 is unknown.

Discussion

Of the 14 Montezuma quail captured, 9 mortalities were attributed to avian predation. The mortality of 1 quail (MQ141) may have been caused by handling stress or exposure to the elements. However, handling stress can be discounted as a cause of mortality, primarily because the other quail were subjected to the same handling procedures and did not show any immediate ill affects. We believe exposure may have contributed to the death of MQ141 for several reasons. First, MQ141 was released alone and did not covey-up. Second, temperatures on the evening of her release fell below 0 C and thermoregulation may have been compromised without roosting with other quail.

The remaining 4 quail mortalities were labeled as mammal ($n = 1$), and miscellaneous ($n = 3$). The miscellaneous category included causes other than predation (e.g., lost signal, transmitter failure, unknown death, etc.).

Previous researchers have documented raptor predation in Montezuma quail (Ligon 1927, Leopold and McCabe 1957, Brown 1982, Stromberg 1990, Holdermann and Holdermann 1993). In fact, Stromberg (1990) suggested that raptors had the tendency to localize their hunting efforts on the same covey.

Although the mortalities in this study may have been natural, other factors may have contributed to the high levels observed. Montezuma quail primarily forage for bulbs and tubers making them extremely difficult to bait into standard funnel traps. Consequently, conventional trapping techniques were useless and we resorted to unconventional methods (i.e. fishing dip nets, mist nets and bird dogs). These unconventional capture techniques were possibly more stressful on the quail. Also, transplanting and releasing away from capture site, releasing in low numbers (i.e. singles or pairs) in addition to being in unfamiliar surroundings perhaps increased their vulnerability to predation. Additionally, unfamiliar surroundings and poor habitat conditions at the release sites, such as the Elephant Mountain site, may have increased their chance of being predated. And lastly, transmitter design potentially restricted movement or interfered with other behavior and contributed to this already stressful arrangement.

Affixing radiotransmitters to quail is a common practice in scientific investigations. Guthery and Lusk (2004) proposed that northern bobwhite are handicapped when radiotransmitters are affixed, thus lowering survival. However, Hernandez et al. (2004) found no difference in body mass, feed consumption, and energy expenditure between collared and un-collared quail. Additionally, Stromberg (1990) noted that no difference was found in life expectancy between radioed and non-radioed Montezuma quail. Although poor attachment can impair mobility and survival of quail, we discount the likelihood that poor attachment played a role in reducing survival of quail in this study. Specifically, the research team had extensive experience in affixing radiotransmitters to 3 quail species and a variety of other gallinaceous birds. Stromberg (1990) suggested that radio attachment on Montezuma quail should be chosen wisely due to their unique foraging behavior.

Another possible explanation for the high mortality documented in this study is that the habitat at the Elephant Mountain release site was in poor condition and did not provide the cover essential for survival. The key component limiting the distribution of Montezuma quail throughout their range is herbaceous cover (Brown 1982). Since they rely on cryptic coloration for concealment and rarely flush, Montezuma quail are especially sensitive to drastic changes in herbaceous cover (Brown 1982, Albers and Gehlbach 1990). In October 2000, Elephant Mountain experienced a drastic freeze that reduced herbaceous cover (S. P. Lerich, Texas Parks and Wildlife Department, unpublished data). In fact, sightings of Montezuma quail were less frequent atop Elephant Mountain following the freeze, suggesting population numbers declined. Because Montezuma quail have limited mobility and defense mechanisms, it is plausible that a reduction in herbaceous cover could increase the vulnerability of quail to avian predation and lead to population reduction. However, this theory does not necessarily apply for the Montezuma quail captured at the Davis Mountain Preserve where cover seemed to be in adequate supply.

Conclusions

Unlike other quail species in North America, Montezuma quail are habitat and foraging specialist. Because of these unique traits, Montezuma quail appear to be susceptible to heavy predation by raptors. Mortality rates may be accelerated if herbaceous cover is reduced to a critical level. The mortality rate of Montezuma quail in this study is similar to that reported by previous researchers (e.g., Stromberg 1990).

The combination of difficulty in capture, stressful capture techniques, moving to and releasing in unfamiliar areas in low numbers, and unconventional capture techniques made studying Montezuma quail extremely challenging. Alternative capture techniques need to be developed and further evaluation of traditional techniques used to affix radio transmitters or transmitter design itself need to be refined if Montezuma quail are to be studied using radio telemetry.

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Over-winter Survival of Northern Bobwhite in Relation to Landscape Composition and Structure

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The conceptualization of security of bobwhite during winter has been predicated on the assumption that winter ranges differ in quality, based on habitat structure, composition, or interspersion. Although some studies have qualitatively related habitat composition to survival, no studies have quantitatively linked habitat or landscape characteristics to winter survival and the specific structural or compositional characteristics that influence quality are unknown. To quantify winter habitat quality, we modeled hazards as a function of habitat characteristics in relation to winter survival of radio-marked bobwhite (2000, n = 118 in 16 coveys; 2001, n = 49 in 7 coveys) in a managed agricultural landscape in Mississippi, as a function of landscape structure and composition at 2 spatial scales (daily and seasonal ranges). For each spatial scale we constructed *a priori* models that estimated year-specific winter survival as a function of unique combinations of variables that characterized landscape composition and structure and had previously been identified as relevant to bobwhite ecology. At the spatial scale of winter ranges, the *a priori* model containing % of landscape, mean patch size, and edge density of linear herbaceous was the best approximating model and suggested a negative effect of linear herbaceous cover on survival. In retrospective analyses, models containing variables describing quantity and structure of linear herbaceous cover and cropland indicated that as these elements increased, risk of mortality increased. At the spatial scale of daily activity, metrics describing landscape structure and composition were poor predictors of survival. During this study, the quantity, patch size, amount of edge, or interspersion of patch types within the winter range or surrounding daily activity locations did not measurably influence the hazard function, suggesting that seasonal ranges can have different composition and structure, yet produce similar survival rates for the birds that inhabit each range.

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Key words: *Colinus virginianus*, landscape composition, landscape structure, northern bobwhite, survival, winter habitat

Introduction

Many studies of animal ecology focus on habitat use relative to availability under the presumption that selective use reflects greater habitat quality or relative value (Garshelis 2000.). These studies often use statistical tests (e.g. chi-square tests, compositional analysis, etc.) to detect disproportionate or non-random use for the purpose of making inferences about habitat preference (Dixon et al. 1996). Although this approach may reflect habitat selection, it does not necessary reflect habitat quality as measured by fitness (Van Horne 1983). Inferences regarding habitat quality require knowledge of relationships among habitat composition, structure, and fitness components (e.g. survival, reproduction).

Habitat has been characterized as the sum of the specific resources, consistent with the behavioral, physiological, and morphological adaptations of a species, required by the individual for reproduction and survival. For bobwhite, habitat quality has been conceptualized under 2 competing models, the quality continuum model (Schroeder 1985, Taylor et al. 1999*b*) and the binary response or usable space model (Guthery 1997). Under the quality continuum model, habitat quality varies from poor to excellent in a continuous fashion. Characteristics of habitat such as thermoregulatory value, energetic resources, or security from predators might

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be expected to vary in a continuous fashion. Under the binary response model, a point within the landscape is either entirely usable or not. That is, some region around a point location provides essential resources required by an individual for survival and reproduction, and thus is usable, or not. Under this model, habitat quality is characterized as the proportion of the landscape usable through time (Guthery 1997). Guthery (1997) suggests that bobwhite density can only be increased by increasing usable space through time and management activities should focus on increasing usable space. In contrast, Taylor et al. (1999*b*) advocated a parameter-based approach to management and suggested that bobwhite management and restoration efforts were likely to be unsuccessful until biologists understood the nature of relationships among management practices, habitat structure, and vital rates that contribute to fitness.

Over-winter survival is an important determinant of bobwhite population performance and may be influenced by the distribution, quantity, and quality of habitats that provide food, thermal cover, and protection from predators. Winter survival has been shown to vary latitudinally (Guthery et al. 2000) and annually in relation to winter weather, snow coverage (Roseberry and Klimstra 1984), harvest rate (Dixon et al. 1996) , ground cover, and predation regimes (Burger et al. 1998). However, despite numerous studies of bobwhite winter habitat use and survival, no studies have linked habitat or landscape characteristics to winter survival. Conroy (1993) suggested that Cox proportional hazard modeling might be used to assess the effects of covariates such as habitat use on fitness measures.

Errington (1935) suggested that as bobwhite populations increased, a greater proportion of individuals occupied marginal ranges and density-dependent mechanisms caused overall population security to decrease. Roseberry and Klimstra (1984) found support for this hypothesis in Illinois in that occupancy rates of winter ranges varied in relation to density. They suggested that high quality ranges were occupied regardless of density, but lower quality ranges were only occupied at high density. Fitness-based, density-dependent habitat selection is predicted by the Fretwell and Lucas (1970) ideal-free habitat selection model. This implies that winter ranges differ in quality, based on habitat structure, composition, or interspersion. However, the specific structural or compositional characteristics that influence quality are unknown.

Some studies have qualitatively related habitat composition to survival (Hines 1987, Klinger et al. 1989, Loegering and Fraser 1995), but few have made quantitative estimations of survival in relation to habitat composition and structure. Landscape structure and composition has been quantitatively related to nesting season (Schmitz and Clark 1999) and winter (Perkins et al. 1997) survival rates for ring-necked pheasants (*Phasianus colchicus*) and nesting season survival for bobwhite (Taylor et al. 1999*a*). However, we could find no reported data relating landscape structure and composition to overwinter survival of bobwhite. To quantify habitat quality, we estimated the influence of landscape structure and composition at 2 spatial scales (seasonal covey range and daily use) on bobwhite survival during winter.

Study Area

This study was conducted on the Black Prairie Wildlife Management Area (BPWMA), in southern Lowndes County, Mississippi, USA. The BPWMA is approximately 2,300 hectares and is owned by the state of Mississippi and managed by the Mississippi Department of Wildlife, Fisheries and Parks (MDWFP). Land cover/land use on BPWMA during the study included: green ash (*Fraxinus pennsylvanica*) successional areas (2.5%), shrubby/herbaceous 16 m wide cover strips in agricultural fields (0.7%), idle mixed exotic and native grasslands (44%), herbaceous 10 m wide field borders around agricultural fields (1.0%), annual food plots (0.7% in 2000, 2.3% in 2001) consisting of clover, wheat, or sunflowers, improved pasture (0.6%), road right-of-ways (0.6%), agricultural row crops (corn-soybean rotation; 27.5% in 2000, 26.1% in 2001), water (1.0%), woodlands (21.2%), and yard areas (<1.0%) surrounding a few

Table 1: List and description of metrics generated in Patch Analyst and FRAGSTSATS to assess effects of landscape composition and structure on over-winter survival of radio-marked northern bobwhite on Black Prairie Wildlife Management Area, Mississippi, 2001-2002.

Metric	Description
SDI	Shannon Diversity Index
ED	Overall Edge Density
MPS	Overall Mean Patch Size
LS_GRASS	% of Range in Grass
ED_GRASS	Edge Density of Grass in Range
MPS_GRAS	Mean Patch Size of Grass Patches in Range
LS_WOOD	% of Range in Woods
ED_WOOD	Edge Density of Woods in Range
MPS_WOOD	Mean Patch Size of Woods Patches in Range
LS_LINH	% of Range in Linear Herbaceous Cover
ED_LINH	Edge Density of Linear Herbaceous Cover in Range
MPS_LINH	Mean Patch Size of Linear Herbaceous Cover Patches in Range
LS_RC	% of Range in Row Crops
ED_RC	Edge Density of Row Crops in Range
MPS_RC	Mean Patch Size of Row Crop Patches in Range

houses and equipment storage facilities on the area. Disturbance, including prescribed fire, disking, and herbicide application was used on BPWMA to maintain early successional herbaceous communities.

Methods

The winter season was defined for this study as September 15 to April 14, 2000-2001 and 2001-2002. Bobwhites were captured during spring and fall of each year. Fall capture took place during 2 weeks in September and 2 weeks in November of each year. Spring trapping began in late January and lasted into March each year. Birds were captured in walk-in style wire traps (Stoddard 1931) baited with cracked corn. Traps were checked twice daily during trapping intervals, once in the morning after birds had foraged and returned to loafing cover and once in the evening after birds had foraged and returned to roost sites. After capture, the gender and age of each bird was determined. Bobwhite age was categorically classified as juveniles or adults. Each animal was then weighed, banded on the right leg with a #7 numbered aluminum band, fitted with a 5-6 g necklace style radio transmitter (American Wildlife Enterprises, Monticello, FL), and released at the capture site. Radio transmitters had a 12-hour mortality sensor, a 20 cm antenna, and operated on a unique frequency between 148.000 and 149.999 MHz. Bobwhite that were caught during the evening trap check when the temperature was less than 10° C or when it was raining were held overnight in wooden boxes measuring approximately 60 cm wide by 60 cm long by 30 cm high with a cloth top and a rubber door flap. Bobwhites that were held overnight were released at the capture site the following morning prior to the morning trap check. Additional bobwhites were captured by night netting throughout the study (Truitt and Dailey 2000).

Radio Location

Radio-marked bobwhites were located at least 5 days/week using a programmable scanning receiver and a handheld 3-element Yagi antenna (Advanced Telemetry Systems, Inc., Isanti, MN). The diel period was divided into 3 time intervals (sunrise to 4 hours post-sunrise, midday, and 4 hours pre-sunset to sunset) and time of location was systematically rotated through these intervals to ensure representative sampling of habitat use. Birds were located by homing in to ≤25 m of the radio signal and circling the location to pin point the exact location of the animal (White and Garrott 1990). Each daily location was georeferenced using a handheld GPS unit (Trimble Navigation Limited, Inc., Sunnyvale, CA). The distance and azimuth from the georeferenced point to the actual location of the bird was recorded. After the data was downloaded from the GPS unit and differentially corrected, the distance and azimuth were used to estimate the location of each bird. Remains of dead bobwhites were recovered upon receiving a mortality signal.

Land Cover Determination

Aerial photographs were georeferenced and imported into Arcview 3.2. Year-specific, vector-based GIS coverages for BPWMA were developed through heads up digitizing of the georeferenced aerial imagery. Individual polygons that were generated were then classified to land cover and ground truthed.

Population Estimation

Fall population size was estimated by counting early morning covey calls (Wellendorf 2000). As a sampling frame for fall density estimation, 87 500 m X 500 m square grid cells were overlayed on the BPWMA GIS coverage. Eighteen of these cells were randomly selected for sampling. Call counts were conducted during the last week of October and the first week of November during each season. Calling rate probabilities were estimated during 10-day periods before and after the call counts using coveys that contained radio marked individuals. Coveys that contained radio marked individuals were located using radio telemetry techniques approximately 1/2 hour before sunrise. When the covey was located and approached to within approximately 35 m, an observer waited until approximately 1/2 hour after sunrise and listened to determine if the covey made early morning covey calls. At approximately 1/2 hour after sunrise, the covey was flushed and the number of individual bobwhites in the covey was counted. Calling rate probabilities were estimated as the proportion of marked coveys that called during the 10-day period preceding and following the covey call counts. Mean covey size was determined by calculating the mean number of individuals in all of the coveys that were flushed. During the call counts, 4 observers were assigned to each of the randomly selected blocks. One observer was placed along each outside edge of the block at the midpoint (250 m from a corner) and faced into the block. When a covey call was heard, the observer recorded the azimuth, approximate distance, and time of the call. Covey locations were verified by coordinating the observed calls, times, and azimuths from all 4 observers for a block and triangulating the estimated locations observed by 2 or more observers. Early fall population size was determined by multiplying the average covey size by the total number of covey calls heard within the sampling blocks and dividing that number by the calling rate probability multiplied by the percentage of the grids that were sampled. The same randomly selected blocks were used during all years.

Covey Associations

Individual radio-marked bobwhites were assigned to coveys according to their capture history and association with other radio-marked bobwhite. Winter covey ranges were estimated using one location/covey/day to generate a 95% kernel seasonal range estimate (Worton 1989). Seasonal range estimates were generated for all coveys that had ≥25 unique locations. Locations for different individuals within the same covey at a given location and time were not considered unique for the purposes of generating seasonal range estimates because of lack of independence associated with the coveying behavior of bobwhite.

Home Range Estimation

To determine the minimum number of unique locations necessary to estimate an accurate seasonal

Table 2: Models used to assess effects of landscape composition and structure on over-winter survival of radio-marked northern bobwhite on Black Prairie Wildlife Management Area, Mississippi 2001-2002 in PROC PHREG.

Model	Metrics Used in Model
Shannon Diversity Index	SDI
Overall Edge Density	ED
Overall Mean Patch Size	MPS
Woody Composition and Structure	LS_WOOD, ED_WOOD, MPS_WOOD
Grass Composition and Structure	LS_GRASS, ED_GRASS, MPS_GRAS
Linear Herbaceous Composition and Structure	LS_LINH, ED_LINH, MPS_LINH
Row Crop Composition and Structure	LS_RC, ED_RC, MPS_RC
% of Range Burned Since Last Growing Season	BURN_F, BURN_S
% of Range Burned and Crass Composition	BURN_F, BURN_S, LS_GRASS

range, we used the 5 coveys from each season with the most locations and used the bootstrap method in Animal Movement extension (Hooge and Eichenlaub 1997) in Arcview 3.2 to create seasonal ranges by randomly selecting 5 samples of locations in increments of 5 locations each from 5-50 without replacement within each set and with replacement between sets (i.e., 5 seasonal ranges with 5 locations each, 5 seasonal ranges with 10 locations each,...,5 seasonal ranges with 50 locations each). Mean estimated range size and SD were plotted against estimated home range size for each covey. Ranges based on 5-20 locations were highly variable. Home range estimates based on >25 locations stabilized within 15% of the mean deviation for all other seasonal ranges consisting of 30-50 locations. We used 25 locations as the minimum number required to estimate seasonal ranges in this study and estimated seasonal range for all coveys that had ≥25 locations. Seasonal ranges were generated for 16 coveys containing 118 radio-marked birds during the 2000-2001 season and 7 coveys containing 49 radio-marked birds during the 2001-2002 season.

Landscape Structure And Composition

We examined relationships among landscape structure and composition and survival at 2 spatial scales. At the coarsest spatial scale, we modeled hazards as a function of time-invariate covariates describing the structure and composition of winter ranges. At a finer spatial scale, we modeled hazards as a function of time-varying covariates that described the immediate landscape context in which individual daily locations occurred.

At the home range scale, we associated habitat characteristics with each seasonal range by clipping the covey range boundary to the BPWMA coverage for the corresponding year. Once habitat attributes were added to each seasonal range, landscape metrics were then generated for each seasonal range using the Patch Analyst extension in Arcview 3.2 (Elkie et al. 1999). Researchers commonly acknowledge that habitat types important to bobwhite include woods, brushy areas, agricultural fields, and grasslands (Stoddard 1931, Leopold 1933, Ridley 1952, Stanford 1952, Edminster 1954). Some researchers have suggested that landscape diversity, interspersion, or composition may influence bobwhite densities (Leopold 1933, Baxter and Wolfe 1972, McRae et al. 1979, Schroeder 1985, Brady et al. 1993). We generated values for 3 landscape-level metrics: Shannon Diversity Index, overall mean patch size, and overall edge density. For each of 4 habitat classes presumed to be important to bobwhite (grass, woods, row crop, and linear herbaceous cover), we generated 3 landscape metrics; i.e., percentage of the landscape, edge density, and mean patch size. In order to avoid problems associated with multicollinearity, we developed candidate models that, within models, included only a single proportional measure of landscape composition. Linear herbaceous cover was composed of field borders and cover strips pooled into one class (Table 1). Additionally, we calculated percentage of the seasonal range that was burned during the fall (BURN_F) and spring (BURN_S) of each season.

Survival Estimation

After generating covey specific landscape metrics, each individual radio-marked bobwhite that was associated with one of the coveys used to generate the seasonal ranges was assigned the 17 landscape metric values corresponding to that bird's covey as covariates. Seasonal survival rates were estimated using the Kaplan-Meier approach modified for staggered entry (Pollock et al. 1989) within PROC PHREG in SAS (SAS Institute, Inc. 1996). Birds with unknown fates (radio-failure, emigration from study area, mortality attributed to research, or survival past April 14th of each year) were right censored. We assumed that right-censoring mechanisms were independent of the bird's fate, left-censored bobwhite had similar survival distributions to birds that were previously included in the risk set, the sample of bobwhite that we used was a random sample from the population of birds on BPWMA, survival times were independent for all individuals, and capture, handling and marking did not affect survival. We modeled hazards as a function of habitat characteristics in SAS using PROC PHREG (Allison 1995) to estimate effect of landscape metrics described above on survival of radio-marked bobwhite. We constructed 9 *a priori* models that estimated year-specific survival as a function of landscape composition and structure using the covariates listed above (Table 2). Additionally, a model that included no covariates was included in the analysis. We controlled for variation between years using a STRATA statement (Allison 1995). Parameter estimates for each covariate included in the above models and Akaike's Information Criterion (AIC) values were generated in SAS using PROC PHREG. Then, we conducted an *a posteriori* analysis in which we modeled the composite models listed above along with each class metric individually, while controlling for variation between years. The model from the set of candidate models with the least AIC value was selected as the best approximating model, given the data and the candidate model set (Table 3).

At the finer spatial and temporal resolution, we modeled hazards as a function of landscape characteristics in the immediate vicinity of daily locations. Daily locations simply reflect a discrete snapshot of the habitat space actually used throughout a given day. In order to better capture the landscape structure within the expected range of daily activities, we buffered daily locations by a radius equal to the mean daily movement observed during this study (156 m). Mean daily movement was estimated as the mean distance between consecutive daily locations for same individual, averaged across individuals. Mean daily movement for the 2000-2001 season was 156.92 m and for the 2001-2002 season was 155.49 m. We used 156 m as the mean daily movement for both years. Because locations were taken only once daily, we assumed that the area within a circle with radius equal to the mean daily movement around daily locations would characterize the region most probably used by that bird throughout that day. We recorded 1,417 daily locations for 79 radio-marked bobwhites during the 2000-2001 season and 2,002 locations for 53 radio-marked bobwhites during the 2001-2002 season.

Each daily location was buffered by 156 m generating a circular daily range polygon. The daily range was then clipped to the BPWMA coverage for the corresponding year in ARC/INFO. The corresponding habitat metrics were calculated for each daily range in FRAGSTATS (McGarigal and Marks 1994). After daily locations had been buffered and clipped, the habitat metrics were added to each record as time-varying covariates, the data were analyzed using PROC PHREG in SAS as described above for

Table 3: Habitat models, parameter estimates, AIC values, and hazard ratios for 2000-2002 over-winter seasonal ranges of radio-marked northern bobwhite on Black Prairie Wildlife Management Area, Mississippi.

Model	Variables	Estimate	SE	Hazard Ratio	AIC	Δ AIC
MPS Linear Herbaceous	MPS_LINH	1.8953	0.8509	6.6540	741.5270	0
Linear Herbaceous Composite [a]	LS_LINH	-0.1255	0.0829	0.8820	741.9020	0.375
	ED_LINH	0.0053	0.0028	1.0050		
	MPS_LINH	2.8606	1.2311	17.4730		
ED Rowcrop	ED_RC	0.0019	0.0010	1.0020	742.6960	1.1690
LS Rowcrop	LS_RC	0.0091	0.0046	1.0090	742.7120	1.1850
ED Linear Herbaceous	ED_LINH	0.0018	0.0010	1.0020	743.1470	1.6200
LS Linear Herbaceous	LS_LINH	0.0412	0.0240	1.0420	743.6580	2.1310
ED [a]	ED	0.0010	0.0007	1.0010	744.3780	2.8510
No Covariate Model [a]					744.4680	2.9410
Grass Composite [a]	LS_GRASS	0.0212	0.0109	1.0210	745.2330	3.7060
	ED_GRASS	-0.0034	0.0017	0.9970		
	MPS_GRAS	-0.3522	0.1977	0.7030		
SDI [a]	SDI	0.4499	0.4131	1.5680	745.2530	3.7260
ED Grass	ED_GRASS	-0.0012	0.0011	0.9990	745.3050	3.7780
MPS [a]	MPS	-0.3362	0.3322	0.7140	745.3920	3.8650
ED Wood	ED_WOOD	-0.0017	0.0020	0.9980	745.7250	4.1980
MPS Grass	MPS_GRAS	-0.0978	0.1209	0.9070	745.7930	4.2660
Rowcrop Composite [a]	LS_RC	0.0090	0.0117	1.0090	746.0110	4.4840
	ED_RC	0.0005	0.0021	1.0010		
	MPS_RC	-0.0439	0.1070	0.9570		
MPS Rowcrop	MPS_RC	0.0314	0.0629	1.0320	746.2260	4.6990
Burn Spring	BURN_S	0.0023	0.0053	1.0020	746.2800	4.7530
MPS Wood	MPS_WOOD	-0.1021	0.2598	0.9030	746.3090	4.7820
Burn Fall	BURN_F	-0.0152	0.0530	0.9850	746.3830	4.8560
LS Grass	LS_GRASS	-0.0004	0.0054	1	746.4610	4.9340
LS Wood	LS_WOOD	0.0006	0.0107	1.0010	746.4640	4.9370
Woody Composite [a]	LS_WOOD	0.0283	0.0218	1.0290	747.7300	6.2030
	ED_WOOD	-0.0033	0.0023	0.9970		
	MPS_WOOD	-0.7497	0.5515	0.4730		
Burn Composite [a]	BURN_F	-0.0089	0.0558	0.9910	748.2540	6.7270
	BURN_S	0.0020	0.0056	1.0020		
Grass + Burn Composite [a]	LS_GRASS	-0.0017	0.0067	0.9980	750.1890	8.6620
	BURN_F	-0.0026	0.0611	0.9970		
	BURN_S	0.0029	0.0067	1.0030		

[a] *a priori* models

seasonal ranges (Table 4). Models that included covariates related to burning were excluded from daily range analysis because they composed a very small portion of each daily range. For this analysis, each location was considered to be an independent observation. Each bird was introduced to the risk set each day with the set of covariates calculated for that day. If the bird survived past that day, it was censored and brought back into the risk set as a new observation on the following day, with a new set of covariates corresponding to the landscape metrics associated with that daily location. If the bird died between that day and the next, it was considered a mortality on that day and that day's landscape metrics were associated with the mortality event.

Mortality Locations

To identify landscape metrics that may have been associated with mortality events, we compared landscape characteristics at locations where dead birds were recovered to known live locations. Mortality and live locations were buffered by 156 m and clipped to land cover layers as described for daily locations. Insofar as recovery sites of dead birds reflect a mixture of actual mortality sites and locations to which depredated birds were translocated and consumed by predators, we did not assume that the location of bird remains was necessarily the location where the mortality occurred. We paired each mortality location with a randomly selected live location for the same bird during the 14 days prior to the recovery of the bird's remains. We used a mixed model ANOVA in SAS using PROC MIXED to compare the habitat composition and structure between live locations and mortality recovery locations. We treated bird ID as a random blocking effect and year and type of location (live or dead) as fixed effects. We compared the 3 landscape metrics, 4 groups of class metrics, and all class metrics individually listed above between live and dead locations.

Results

Population Estimate

The early fall population size on Black Prairie Wildlife Management Area was estimated as 1,849 (SE = 1,170.9) individuals for the 2000-2001 season and 891 (SE = 1,140.4) individuals for the 2001-2002 season. Of the estimated early fall population, we radio-marked 9.1% (n = 169) during the 2000-2001 season and 7.9% (n = 70) during the 2001-2002 season.

Winter Survival

Over-winter survival differed dramatically between years. Survival from 15 September-14 April was 0.060 (SE = 0.019) during the 2000-2001 season and 0.465 (SE = 0.110) during the 2001-2002 season. During the 2000 growing season, this study site experienced a severe drought and ground cover conditions were poor going into winter (Holt 2003). During the 2000-2001 winter season this population experienced high avian cause-specific mortality (Holt 2003).

Covey Range Size

Covey range sizes also differed between years. Mean winter covey range size was 22.46 ha (n = 16, SE = 14.4) during the 2000-2001 season and 44.13 ha (n = 7, SE = 45.1) during the 2001-2002 season. The combined area occupied by radio-marked coveys was 320.23 ha during the 2000-2001 season and 275.44 ha during the 2001-2002 season. Thus radio-marked coveys occupied approximately 13.9% of the total study area during the 2000-2001 season and 12.0% during the 2001-2002 season. The area of the overlapping portion between seasons was 67.01 ha. Twenty-one percent of the 2000-2001 cumulative range was used by radio-marked bobwhite during the 2001-2002 season, and 24% of the 2001-2002 range had been used by radio-marked bobwhite during the 2000-2001 season. Thus, radio-marked bobwhite occupied similar proportions, but different regions, of the total study area between years. Insofar as the entire study area was systematically trapped in both years, differences in occupied areas more likely reflect annual differences in space use rather than spatial distribution of sampling effort.

Covey Range Landscape Metrics

Of the 10 *a priori* models, the linear herbaceous composite model was selected as the best approximating model with an AIC 2.476 less than the next best model (ED model) and 2.566 less than the no covariates model (Table 3). The linear herbaceous model included variables describing % of the landscape in linear herbaceous cover (LS_LINH, β = -0.00192, SE = 0.08292), edge density of linear herba-

Table 4: Habitat models, parameter estimates, AIC values, and hazard ratios for 2000-2002 over-winter daily ranges of radio-marked northern bobwhite on Black Prairie Wildlife Management Area, Mississippi.

Model	Variable	Estimate	SE	Hazard Ratio	AIC	Δ AIC
LS Linear Hebaceous Cover	LS_LINH	0.0373	0.0218	1.0380	356.5160	0
MPS Linear Herbaceous Cover	MPS_LINH	0.6485	0.3770	1.9130	356.6920	0.1760
No covariates [a]					357.1380	0.6220
ED Linear Herbaceous Cover	ED_LINH	0.0030	0.0022	1.0030	357.2920	0.7760
SDI [a]	SDI	-0.4251	0.3678	0.6540	357.8150	1.2990
Distance to Woody Cover	DIS_WOOD	0.0019	0.0021	1.0020	358.3530	1.8370
ED Rowcrop	ED_RC	0.0013	0.0018	1.0010	358.6050	2.0890
LS Rowcrop	LS_RC	0.0032	0.0045	1.0030	358.6390	2.1230
ED Rowcrop	ED_WOOD	-0.0013	0.0021	0.9990	358.7160	2.2000
ED [a]	ED	0.0006	0.0016	1.0010	358.9950	2.4790
MPS Grass	MPS_GRAS	0.0253	0.0759	1.0260	359.0280	2.5120
LS Wood	LS_WOOD	0.0023	0.0087	1.0020	359.0660	2.5500
LS Grass	LS_GRASS	0.0011	0.0046	1.0010	359.0840	2.5680
MPS [a]	MPS	0.0285	0.1469	1.0290	359.1010	2.5850
MPS Wood	MPS_WOOD	0.0234	0.1280	1.0240	359.1050	2.5890
ED Grass	ED_GRASS	-0.0001	0.0017	1	359.1370	2.6210
MPS Rowcrop	MPS_RC	0.0010	0.0988	1.0010	359.1380	2.6220
Linear Herbaceous Composite [a]	LS_LINH	0.0610	0.1146	1.0630	360.3160	3.8000
	ED_LINH	-0.0026	0.0075	0.9970	360.3160	
	MPS_LINH	0.0147	1.1081	1.0150	360.3160	
RowCrop Composite [a]	LS_RC	0.0088	0.0120	1.0090	361.9220	5.4060
	ED_RC	-0.0004	0.0036	1	361.9220	
	MPS_RC	-0.1405	0.1765	0.8690	361.9220	
Woody Composite [a]	LS_WOOD	0.0101	0.0176	1.0100	362.3180	5.8020
	ED_WOOD	-0.0020	0.0023	0.9980	362.3180	
	MPS_WOOD	-0.0861	0.2480	0.9180	362.3180	
Grass Composite [a]	LS_GRASS	0.0002	0.0103	1	363.0250	6.5090
	ED_GRASS	-0.0001	0.0025	1	363.0250	
	MPS_GRAS	0.0232	0.1450	1.0230	363.0250	

[a] *a priori* models

ceous cover (ED_LINH, β = 0.00525, SE = 0.00284), and mean patch size of linear herbaceous cover (MPS_LINH, β = 2.86064, SE = 1.23112). The confidence intervals on the parameter estimates for LS_LINH and ED_LINH included 0, and the sign suggested a weak positive and negative effect, respectively, of these variables on winter survival. The confidence intervals on mean patch size of linear herbaceous cover did not include 0 and the sign and hazard ratio indicated that as mean patch size of linear herbaceous cover in the range increased, risk of mortality increased. In retrospective analyses that included all 10 *a priori* models + 14 single variable models, 5 models, including the linear herbaceous composite model, had ΔAIC < 2 and therefore were considered as competing models (Table 3). Single variable models that included mean patch size of linear herbaceous cover, edge density of rowcrop, %

of landscape in rowcrop, and edge density of linear herbaceous cover all indicated that as the landscape metric increased, risk of mortality increased (Table 3).

Daily Location Landscape Metrics

Of the 8 *a priori* models estimating survival as a function of characteristics of daily ranges, the no covariates models had the lowest AIC, however, 3 other models (SDI, ED, MPS) were within 2 ΔAIC of the best approximating model, suggesting considerable model uncertainty (Table 4). Additionally, confidence intervals on parameter estimates included 0 for all competing models, providing little evidence for substantive effect on survival of landscape structure and composition within daily activity regions. In the retrospective candidate model set, the percentage of landscape in linear herbaceous cover was the best approximating model, but 5 other models, including the no covariates model, were competing (ΔAIC < 2). Confidence intervals for coefficients of all variables in all competing models included 0.

Mortality Recovery Locations

Live locations did not differ ($P > 0.05$) from mortality locations, regarding overall mean patch size, overall edge density, percentage of the landscape in linear herbaceous cover, percentage of the landscape in grass, edge density of woods, edge density of linear herbaceous cover, edge density of grass, mean patch size of woods, mean patch size of linear herbaceous cover, mean patch size of grass, mean patch size of row crop, or distance to wooded edge. We observed a year by location type interaction for Shannon diversity index ($F_{1,68}$ = 9.66, P = 0.0027), percentage of the landscape in wooded cover ($F_{1,68}$ = 4.23, P = 0.0437), percentage of the landscape in row crop ($F_{1,68}$ = 4.24, P = 0.0433), and edge density of row crops ($F_{1,68}$ = 9.67, P = 0.0027). During the second season, Shannon diversity index differed between mortality and live locations ($F_{1,68}$ = 9.56, P = 0.0029), with mortality locations occurring in more diverse landscapes (SDI = 1.0592, SE = 0.09025) than live locations (SDI = 0.8015, SE = 0.09025). Shannon diversity index at mortality locations differed between years ($F_{1,68}$ = 4.00, P = 0.0494), with less diverse locations during the 2000-2001 season (SDI = 0.8591, SE = 0.04310) than during the 2001-2002 season (SDI = 1.0592, SE = 0.09025). During the 2000-2001 season, mean percentage of the landscape in wooded cover differed ($F_{1,68}$ = 4.10, P = 0.0468) between mortality (20.65%, SE = 2.04) and live locations (15.06%, SE = 2.4309). The percentage of the landscape in row crop differed ($F_{1,68}$ = 4.35, P = 0.0407) between live (15.47%, SE = 8.2988) and mortality (31.83%, SE = 8.2988) locations during the 2001-2002 season. Similarly, during the 2001-2002 season, mortality locations had greater ($F_{1,68}$ = 9.15, P = 0.0035) edge density of row crops (96.02 m rowcrop/100 ha, SE = 21.4984) than live locations (37.87 m of row crop edge/100 ha, SE = 21.4984).

Discussion

Roseberry and Klimstra (1984) observed that, in Illinois, annual occupancy rates of specific winter ranges differed among ranges and varied from 27-80%. They suggested that these differences might be a function of varying habitat quality, or in the terms of Errington and Hamerstrom (1936), “security”. It has commonly been assumed that habitat quality, or security, does vary among covey ranges, with high occupancy reflecting high quality ranges. It follows that only the most optimal ranges will be occupied at low densities and as density increases, increasingly marginal ranges will become occupied. If winter ranges differ in quality, and bobwhite exhibit ideal free habitat selection (Fretwell and Lucas 1970), then at high population densities, a greater proportion of coveys must inhabit ranges of poorer quality than they would at lesser population densities. If so, this provides a natural mechanism for density-dependent winter mortality observed by Roseberry and Klimstra (1984) on their Illinois study site. Negative correlations between percentage of ranges occupied and population densities have been taken as evidence to support the hypothesis “that individual and collective security declines as population density increases” (Roseberry and Klimstra 1984, p. 30). This hypothesis is predicated on the assumption that survival varies among covey ranges and that security within a covey range (range-specific survival rate) is a function of habitat characteristics.

We studied a bobwhite population in Mississippi during 2 years with dramatically differing density and winter survival. Under the Roseberry and Klimstra (1984) hypothesis, we would predict that a high proportion of the ranges occupied during the low density year (2001-2002) would have been occupied in the high density year (2000-2002) and a low proportion of the ranges occupied during the high density year would be occupied during the low density year. However, we observed that only 24% of the range used during the low density year (2001-2002) had been occupied during the high density year (2000-2001) and a similar proportion (21%) of the area occupied during low density year was occupied at high density. Therefore, in regard to occupancy and density relationships, our observations did not support the quality/security/density hypothesis.

A second prediction of the quality/security hypothesis is that ranges differ in habitat quality, presumably attributable to differences in vegetation structure, patch characteristics, landscape composition, or landscape structure. We found only weak evidence to support the hypothesis that variation in survival of individual birds was related to variation in landscape structure or composition of winter ranges or that survival varies between covey ranges in relation to landscape structure. Finally, our study was conducted at a relatively southern latitude, whereas Roseberry and Klimstra's work was conducted at a more northern latitude with more severe winter weather. If factors that vary among winter ranges, and influence quality, relate to thermal cover or vegetation structure in the presence of snow and ice, differences in response between northern and southern landscapes would be expected. However, we did not find support for dramatic range-specific differences in landscape structure and composition that contribute to variation in winter survival. Several essential habitat components of winter covey ranges have been identified by various researchers. A brushy or woody "headquarters" area for midday loafing, escape cover, and foul weather roosting seems to be an essential habitat component of winter ranges (Stoddard 1931, Roseberry 1964, Bartholomew 1967, Yoho and Dimmick 1972, Roseberry and Klimstra 1984). An adequate and accessible food resource, including cultivated crops and annual weeds associated with cultivation, is essential and the availability or distribution may affect over-winter survival (Roseberry and Klimstra 1984, p. 31). Various early successional stages of grassy or weedy vegetation may provide essential early season roosting and feeding habitats. The importance of quality, quantity, and distribution of these cover types has been recognized (Edminster 1954, Schroeder 1985, Roseberry and Klimstra 1984). The essential nature of interspersion and juxtaposition of these resources to allow simultaneous access to habitat components that meet daily requirements has been recognized (Stoddard 1931, Leopold 1933, Roseberry and Klimstra 1984) and measures of interspersion have been used to quantify "habitat quality" (Baxter and Wolfe 1972, Schroeder 1985). However, relationships among habitat quantity, spatial distribution, and over-winter survival have not been examined for bobwhite. During our study, we observed no consistent relationships among landscape composition/structure and survival at either the spatial scale of the winter range or daily location. Specifically, the quantity, patch size, amount of edge, or interspersion of patch types within the winter range or surrounding daily activity locations did not measurably influence the hazard function, suggesting that seasonal ranges can have different composition and structure, yet show similar survival rates for the birds that inhabit each range. There was no strong evidence to suggest that seasonal ranges differed quantitatively in their survival benefits in relation to the composition or structure of the habitat within them.

Guthery (1999) defined usable space within different arrangements of the habitat as slack and gave 3 reasons why this may occur: (1) bobwhite are adapted to a range of habitats, (2) bobwhite may change the time that they spend on different activities, and (3) different types of patches may serve similar functions. Our observations are consistent with predictions of the Guthery (1999) slack hypothesis, that is different configurations of patch types result in comparable fitness. Guthery et al. (2001) similarly concluded that landscape composition was more important in determining bobwhite abundance than configuration.

Covey range models that best explained survival included all models that used linear herbaceous components, in the composite model and individually. The grassy composite model also was included in the top covey range models. This would suggest that linear herbaceous cover and grass composition and structure may influence bobwhite overwinter survival. In general, there is weak evidence to suggest that some components of linear herbaceous cover have a negative influence on bobwhite over-winter survival. However, due to the placement of cover strips and field borders, this may be an effect of row crops closely associated with these components of the landscape. The row crops at this time of year are harvested and the ground is generally bare or sparsely vegetated for a large portion of the season. Thus, birds occupying covey ranges with significant amounts of linear herbaceous cover might spend a disproportionate amount of time foraging in a high risk environment.

Guthery (1997) proposed the concept of space-time saturation in habitat management for bobwhite. He suggested that a point on the landscape is either usable or not, and as such, managers should strive to provide usable points at all locations at all times throughout the year. The data presented here would lend support to this hypothesis as well. Each point that was recorded for a radio-marked bobwhite was compared to all other points taken for radio-marked bobwhite. There was no strong evidence to indicate that the composition or structure of the habitat surrounding those points strongly influenced survival of bobwhite.

At the point scale, models that best explained survival in relation to habitat composition and structure were ones that included linear herbaceous cover, grass cover, and woody cover individually. Shannon Diversity index was also included in the top daily habitat models. The only individual woody cover model that was not included in the top models (ΔAIC < 2) was mean patch size of woody cover (ΔAIC $= 2.018$). However, the confidence intervals on coefficients included 0, providing relatively weak evidence for influence on survival. This would lend support to the theory that all of the points where data were collected were in usable space at the time they were collected. However, because this data were taken using radio locations of marked bobwhite, one would expect that the points used in this analysis were all at usable locations. Through radio tracking data, we can only see points that are used. Points not used will not be included because a non-usable point will not have a radio location associated with it.

Even though there was weak evidence to suggest influence of these habitat components on survival (i.e. confidence intervals on coefficients included 0), the individual linear herbaceous cover models showed negative influence on survival in the same pattern as those for covey ranges. Once again, as mentioned above, this may be an artifact of the close association of linear herbaceous cover and row crops during this time of year. Similarly, all individual grassy cover models showed weak negative influence on survival. As the percentage of landscape in woody cover and the mean patch size of woody cover increased, there was weak evidence for a decrease in survival. Also, there was a small increase in survival associated with an increase in the edge density of woody cover and a small decrease in survival as distance to woods increased. This would suggest that bobwhite require some woods in the landscape to provide escape cover, but not large blocks of woods.

Although composition and structure within winter ranges and at daily locations were poor predictors of survival, landscape context did differ between live locations and presumed mortality locations. Most notably, during the season with greater survival (2001-2002), mortality locations occurred in landscapes with more row crop (31.83%) than that surrounding live locations (15.47%) and greater row crop edge density (96.06 m/100 ha vs. 37.87 m/100 ha, respectively). Previous research has found that bobwhite use row crop fields less than other types of habitat in their range during winter (Yoho 1970). During winter, row crop fields are usually harvested and provide little to no overhead or vertical cover to protect bobwhite against predation. If row crop fields and in particular edges within row crop fields are used as travel lanes for predators, the predator may go out into other areas to catch prey and then return with the carcass to a spot that it normally uses to consume its prey.

We have attempted to assess habitat quality in this study with a demographic design. It has been suggested that the demographic approach to habitat quality studies is superior to other designs (Garshelis 2000., Garton et al. 2001). We found no strong evidence to suggest that the habitat composition or structure strongly influences bobwhite survival at either the level of the seasonal range or the daily range. This could occur on an area that has already reached space-time saturation (Guthery 1997). This may not be the case on BPWMA and without having unusable points in time and space to compare to usable points, it is impossible to quantify all points on the area. What may be necessary is a synthesis of the demographic design and a more traditional used point versus random point comparison. This could be accomplished by taking all points that were used, buffering them by the mean daily movement, combining the buffered areas, and removing the resulting area from the GIS coverage as usable space, then an equal number of random points could be placed on the remaining coverage and analyzed in the same fashion as the used points. This would give a way to quantify used versus unused portions of the area.

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An Evaluation of Short-term Mesocarnivore Control for Increasing Hatch Rate in Northern Bobwhites

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We evaluated the efficacy of short-term trapping on scent-station visitation rates for some nest predators and survival of artificial nests with chicken eggs at 4 sites in west Texas from 1998-2001. Trapping of predators was conducted with cage traps for 30 days just prior to nest initiation (mid-May through mid-June) at a trap density of 1 trap/20 ha. Each site included a treatment (trapped) and control (non-trapped) area that comprised approximately 250 ha. Scent stations were employed before and after trapping to assess impacts of trapping on predator activity/abundance. Simulated nests (using 3 chicken eggs) were established 1-2 days after trapping ended, and monitored weekly to estimate visitation rate. We removed an average of 69 mesomammals per year ($n = 274$ across all sites), within a 30-day-trapping period. We detected no consistent declines in scent-station visitation rates of target species before or after trapping. We did not detect an increase in survival of artificial nests. We conclude that short-term trapping efforts on small areas used in this study did not reduce the overall predator community enough to affect scent-station visitation rates or survival of artificial nests.

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Key words: *Colinus virginianus*, mesomammal abundance, nesting success, northern bobwhite, predator control, Rolling Plains, Texas

Introduction

Northern bobwhite (*Colinus virginianus*, hereafter bobwhite) populations have declined over most of their historic range since 1980 (Sauer et al. 2001) and at an average of 4.9% annually since 1981 in Texas. Although reasons for the decline are unclear, factors implicated in this decline include habitat fragmentation, land-use changes, weather patterns, fire ants (*Solonopsis invicta*), and an increase in mesomammal predators (Rollins and Carroll 2001, Rollins 2002, Brennan et al. 2005).

Nest survival of bobwhites reported in the literature averaged 28% (Rollins and Carroll 2001) and occasionally nesting success is <20% because of depredation (Lehmann 1984, Ortega et al. 1998, Sloan et al. 1998, Clawson and Rotella 1998). Given the impact of predation on quail survival and nesting success, it seems logical that intensive predator control would increase quail density. However, previous research in South Texas demonstrated that intensive year-long predator control had little effect on quail survival and nesting success (Beasom 1974, Guthery and Beasom 1977).

Given the potential impacts of nest predation by mesomammals (Rollins and Carroll 2001), we tested the efficacy of short-term-selective-predator trapping immediately before nesting season as a means of increasing survival of simulated nests. Predator control efforts usually have little impact on long-term mesomammal densities (Balser et al. 1968). However, intensive predator control efforts immediately before nesting season may offer a window of opportunity for quail to nest with reduced disturbance from predators. We assessed survival of artificial nests on trapped and non-trapped areas and changes in predator populations following short-term control efforts.

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Study Area

Study 1. 1998-1999.

This study was conducted in Tom Green and Coke counties, Texas. This study was conducted using 2 sites, each with 2 study plots (trapped and non-trapped) in 1998 and then repeated with new study sites in 1999. Each study plot was approximately 250 ha and was separated by $\geq$3.0 km.

Site 1 (Angelo State University Management, Instruction, and Research Center) was located approximately 6 km north of San Angelo in Tom Green County, Texas. Site 2 (North Concho Ranch) was located approximately 13 km north of San Angelo in Tom Green County, Texas. Site 3 (Johnson Ranch) was located approximately 6 km northwest of Water Valley in Coke County, Texas. Site 4 (Clark and D Cross ranches) was located approximately 4 km northwest of Water Valley in Coke and Tom Green counties, Texas. All sites were rangeland dominated by an overstory of honey mesquite (*Prosopis glandulosa* var. *glandulosa*) and understory of mixed grasses [mostly tobosa grass (*Hilaria mutica*)], curly mesquite (*Hilaria belangeri*), and prickly pear cactus (*Opuntia* spp.). Thirty-year normals for Coke and Tom Green counties indicate that Study 1 was completed during a drought. Annual precipitation was 32.9 cm in 1998, in Tom Green County and 34.3 cm in 1999, as reported by the National Weather Service at Mathis Field, San Angelo, Texas. Precipitation for 1999 in Coke County was 36.2 cm as reported by the National Weather Service at Water Valley, Texas.

Study 2. 2000-2001.

The second study was initiated in Parker County, Texas, in 2000 and replicated in Coleman County, Texas in 2001.

Parker County - Site 1 was enclosed within a 2.0-m chain-link "game-proof" fence. This pasture is approximately 600 ha and was used for grazing by cattle and exotic ungulates. Study plot 1 of site 2 was located in a 1,900-ha pasture used solely for cattle grazing. Study plot 2 of site 2 was located on a separate portion of the ranch about 20 km southwest of the main ranch property. The treatment area was in the middle of a pasture approximately 800 ha in size. Trapped areas were separated from non-trapped areas by 2.0 km. Sand plum (*Prunus gracilis*), little bluestem (*Schizacharium scoparium*), and mesquite were common. Pecan (*Carya illinoensis*) and live oak (*Quercus virginiana*) dominated the riparian lowlands. Annual precipitation was 95.3 cm in 2000, and was 11.7 cm above 30-year normals as reported by the National Weather Service at Weatherford, Texas.

Coleman County - All sites were grazed by cattle and dominated by mesquite and prickly pear. As a result of drought, low numbers of bunchgrasses were present, and most areas in 2001 were dominated by annual forbs such as broomweed (*Xanthocephalum dracunculoides*), and basketflower (*Centaurea americana*), and annual grasses (*Bromus* spp. and *Hordeum pusillum*). Annual precipitation was 69.9 cm in 2001, as reported by the National Weather Service in Coleman, Texas.

Methods

Mesomammal Control

We targeted specific mesocarnivores such as raccoons (*Procyon lotor*) and skunks (*Mephitis mephitis*) for removal using the most humane means available (e.g., cage traps). Coyotes (*Canis latrans*) and bobcats (*Lynx rufus*) were not targeted because their home range encompassed areas greater than the study sites. Trapping methods also did not facilitate the capture of larger carnivores. Trapping methods were constrained by landowner requirements. For example, spotlighting or calling for predators, such as skunks and coyotes was not allowed.

Trapping was initiated just prior to the nesting season of bobwhites (i.e., late-April) and continued for 30 days. In 2000, trapping was extended 4 days as a result of flooding. Cage traps (36 cm x 36 cm x 122 cm) were placed along pasture roads across each trapped area at a density of approximately 1 trap/20 ha. Chicken eggs were used as bait for the first 15-day period of trapping, while canned cat food was used during the second 15-day period. Traps were checked daily between 0700-1000 hours. Mesomammals were sacrificed with a .22 rifle to the base of the skull under an animal use protocol approved by the Care of Animal Laboratory and Wildlife Use Committee at Texas A&M University (SPR-0198-922). Specimens were deposited in the Angelo State Natural History Collection (ASNHC) as voucher specimens for Tom Green, Coke, Parker and Coleman counties, Texas.

Mesomammal Abundance/Activity

We estimated mesomammal abundance/activity with scent station surveys before trapping and at 2 and 120 days post-trapping on all sites (Woods 1959). Scent stations consisted of a 1.0-m circle of crushed lime on bare ground placed on alternating sides of the road at 0.5-km intervals (Linhart and Knowlton 1975, Martin and Fagre. 1988). Previous studies suggested that movements by coyotes along roads are typically less than 0.5 km (Hodges 1975, Roughton and Sweeny 1982). Given that ranges of coyotes are greater than that of targeted species (i.e., raccoons, skunks, opossums), we feel that distances between stations are sufficient for independence. A cotton swab soaked in bobcat urine was placed in the middle of the plot as a scent attractant (Morrison 1981, Carroll et al. 1999). Bobcat urine was used as a general attractant for multiple species (Conner et al. 1983). Any bias due to the scent attractant used was assumed to be the same across all sites. Each station was observed for 2 consecutive nights (Roughton and Sweeny 1982). Lime and scent attractant were refreshed as necessary.

For these studies, each area had a different number of stations because the length of roads differed among study areas. Furthermore, all study areas contained only a single road that traversed the area. As a result, each area consisted of only 1 transect with a different number of stations. To eliminate differences is sample size of scent stations, visitation rates of targeted species were standardized to 100-scent station nights (SSN).

Simulated Nests

The selection of nesting substrate for simulated nests was restricted to suitable nest clumps meeting the definitions by Lehmann (1984). A prickly pear was deemed suitable for nesting if it was >1 m in size and had grass growing inside the plant (Slater et al. 2001). Three medium-sized, unwashed chicken eggs were placed inside each nest. In order to alleviate bias from human scent at the nest site, latex gloves were worn while constructing the nest and handling the eggs. A 2.0-cm steel washer was placed in the bottom of the nest to facilitate location of the nest site in the event all eggs were removed by predators. Nests were monitored every 7 d for 28 d. Any nest with 1 or more egg(s) missing or moved >40 cm was classified as depredated. After 2 weeks, eggs in surviving nests were replaced to minimize scent resulting from rotting.

No attempt was made to identify individual nest predators. Staller et al. (2005) indicated that diagnostic sign is unreliable for identifying predators of bobwhite nests. Some studies also have shown that predators often leave similar remains suggesting that identification of predators from eggshell remains is equivocal (Hernandez et al. 1997, Marini and Melo 1998, Lariviere 1999). Simulated nests were constructed after post-trap scent stations were completed to measure rates of nest depredation on all sites. The number of transects and placement of nests varied slightly across studies.

Study 1. 1998-1999 - Six transects of 12 simulated nests/transect were placed on each treatment area (Ortega et al. 1998). Transects were 600 m long and placed 200 m apart. Transects were randomly located throughout each study site, and placed perpendicular to an existing fence or road. Nests were placed every 50 m along the transect line and 10 m either to the right or left of the transect line. Nests were located alternately in either a suitable bunchgrass or a clump of prickly pear.

Study 2. 2000-2001 - Eight transects of 6 nests each were established at each site. Transects were approximately 300 m long, and transect lines were approximately 200 m apart. In 2000, nests were constructed in suitable bunchgrass cover ≥10 m off the transect line. As a result of low numbers of suitable bunchgrass clumps in 2001, nests were placed in the nearest clump of grass or prickly pear resembling suitable nesting cover.

Vegetation Sampling

Potential nesting sites were estimated using a belt transect on all sites in order to assess the similarities or differences in vegetative communities among study areas. Simulated nest transects (described above) served as transects for the estimates. Each transect was walked with the observer's arms outstretched and suitable nest sites that were rooted within this belt were counted (Slater et al. 2001). The number of potential grass and prickly pear nests were counted and converted to nests/ha.

Statistical Analyses

Simulated nest survival was analyzed using 2-way Analysis of variance (ANOVA). Sites were pooled and data were analyzed by treatment and year. Linear regression analysis was used to determine if relationships between survival of simulated nests and nest site availability exist. Logistic regression was used to determine if nest site abundance was a good predictor of survival of artificial nests. Sites were pooled and each year was analyzed independently. All statistical analyses were calculated with SPSS Inc. (SPSS Inc. 2002).

Results

Mesomammal Control

1998 - A total of 120 mesomammals was removed; 66 from trapped study plot of site 1, and 54 from the trapped study plot of site 2. Raccoons were the species trapped most frequently on both sites (36%, *n* = 24 on Site 1; 57%, *n* = 31 on Site 2). Other species trapped on Site 1 included opossum (*Didelphis virginianus*, *n* = 16), striped skunk (*n* = 16), and additional mammals (*n* = 8). Other species trapped on Site 2 included opossum (*n* = 10), striped skunk (*n* = 9), and additional mammals (*n* = 4).

1999 - A total of 99 mesomammals was removed; 27 from the trapped study plot of site 3, and 72 from the trapped study plot of site 4. Raccoons were the species most frequently trapped on Site 3 and Site 4 (66%, *n* = 18; 72%, *n* = 52, respectively). Other species trapped on Site 3 included striped skunk (*n* = 6), and miscellaneous mammals (*n* = 3). Other species trapped on Site 4 included striped skunk (*n* = 7), and miscellaneous mammals (*n* = 13).

2000 - A total of 41 mesomammals was removed; 30 from the trapped study plot of site 1, and 11 from the trapped study plot of site 2. Raccoons were the species trapped most frequently on site 1 (*n* = 20). Other species included armadillos, (*Dasypus novemcinctus*, *n* = 3) and opossums (*n* = 7). Armadillos were trapped most frequently on site 2 (*n* = 6), followed by opossums (*n* = 4), and raccoons (*n* = 1). All predators were removed from the riparian areas of the study plots. Predator trapping in the open grasslands proved unsuccessful. No skunks were trapped on either site.

2001 - Predator removal was less successful with only 14 mesomammals removed from the study site; 6 from study plot 1 and 8 from study plot 2. Raccoons were the only species trapped on site 1. Raccoons and skunks were trapped from site 2, but in low numbers (4 of each species). All predators were taken from traps located within close proximity of water (e.g., stock tanks).

Mesomammal Abundance

1998 - Scent stations did not show decreases in number of mesomammal visitations after trapping (Table 1). Further increases in visitations by target species were demonstrated with 120-day post trap scent stations. Raccoons increased from 5.6 and 5.8 visits/100 SSN to 39.5 and 11.6 visits/100 SSN on sites 1 and 2, respectively.

1999 - While numbers of predator visitations decreased overall on scent stations after trapping on both sites 3 and 4, there were no consistent declines in predator abundances on trapped areas. Where 1 species, such as raccoons decreased, skunks may have increased. As in 1998, predator numbers rebounded to or exceeded pre-trap levels by 120-day post trap scent stations.

2000 - Scent stations indicated a decrease in the number of targeted mesomammal visitations after trapping on site 5; however, visitations increased on

Table 1: Pre- and 2-day post-trap scent station visits for targeted species [standardized to visits/100 scent station nights (SSN)] and number of mesomammals removed on respective sites are reported for 8 treatment sites in the Rolling Plains of Texas, 1998 - 2001.

Year	Site	Treatment	County	Scent station visits/100 SSN Pre-trap	 2-day Post trap	Mesomammal trapping Animals removed
1998	1	Trapped	Tom Green	13.9	17.5	66
	2	Trapped		15.0	18.8	54
1999	3	Trapped	Tom Green/Coke	35.3	26.9	27
	4	Trapped		40.7	38.1	72
2000	5	Trapped	Parker	45.4	12.5	30
	6	Trapped		10.0	23.5	11
2001	7	Trapped	Coleman	63.3	33.3	6
	8	Trapped		24.1	13.3	8

site 6 (Table 1). The result of low trapping numbers of some target species (e.g., raccoons, opossums) and the inability to remove others (i.e., skunks) suggest that values showing decreases in mesocarnivore abundances may be misleading (Table 1).

2001 - Scent station visitations suggested that raccoons and skunks were the major species present. There were decreases in visitations on both trapped sites; however, trapping results were so low (Table 1) that post-trap scent station values were not representative of the mesomammal community.

Simulated Nest Survival and Suitable Nest Site Availability

1998 - Average percent of artificial nests that survived 28 days was lower on control versus treatment sites (control [n = 12, $\bar{x}$ = 2.75, SD = 5.44], treatment [n = 12, $\bar{x}$ = 9.58, SD = 12.14]). However, trapping of specific mesomammals did not significantly increase survival of artificial nests (F = 3.165, df = 1, P = 0.089). Greater nest site availability (i.e., suitable nest sites/ha) on control ($\bar{x}$ = 983, SE = 165) versus treatment areas ($\bar{x}$ = 716, SE = 129) suggested a possible relationship between artificial nest survival and nest site availability. A linear relationship between survival of artificial nests and number of suitable nest sites was indicated (F = 4.657, df = 1, P = 0.042) although little variation in artificial nest survival was explained by nest site availability (R^2 = 0.175). Logistic regression was used to determine if nest site abundance was a good predictor of artificial net survival. No relationship between nest site abundance and survival of artificial nests was found on either control (X^2 = 0.017, df = 1, P = 0.897) nor treatment sites (X^2 = 0.142, df = 1, P = 0.707) thus suggesting that nest site abundance is not a good predictor of artificial nest survival.

1999 - Average percent of artificial nests that survived 28 days was higher on control versus treatment sites (control [n = 12, $\bar{x}$ = 43.33, SD = 26.96], treatment [n = 12, $\bar{x}$ = 36.75, SD = 23.27]). Trapping of specific mesomammals did not significantly increase survival of artificial nests (F = 0.410, df = 1, P = 0.529). Greater nest site availability (i.e. suitable nest sites/ha) on control ($\bar{x}$ = 809, SE = 136) versus treatment areas ($\bar{x}$ = 728, SE = 139) suggested a possible relationship between artificial nest survival and nest site availability. A linear relationship between survival of artificial nests and number of suitable nest sites was indicated (F = 6.311, df = 1, P = 0.020) although little variation in artificial nest survival was explained by nest site availability (R^2 = 0.223). Logistic regression was used to determine if nest site abundance was a good predictor of artificial net survival. No relationship between nest site abundance and survival of artificial nests was found on the control (X^2 = 1.56, df = 1, P = 0.211) site. However, a significant relationship was observed on the treatment (X^2 = 6.57, df = 1, P = 0.010) site suggesting that nest site abundance can be a predictor of artificial nest survival.

2000 - Average percent of artificial nests that survived 28 days was higher on control versus treatment sites (control [n = 16, $\bar{x}$ = 73.96, SD = 22.73], treatment [n = 16, $\bar{x}$ = 18.76, SD = 24.24]). Trapping of specific mesomammals had a significant effect on survival of artificial nests, however, trapping did not cause an increase in the survival of artificial nests (F = 44.156, df = 1, P = 0.000). Greater nest site availability (i.e., suitable nest sites/ha) on control ($\bar{x}$ = 2262, SE = 271) versus treatment areas ($\bar{x}$ = 930, SE = 187) suggested a possible relationship between artificial nest survival and nest site availability. A linear relationship between survival of artificial nests and number of suitable nest sites was indicated (F = 4.184, df = 1, P = 0.050) although little variation in artificial nest survival was explained by nest site availability (R^2 = 0.122). Logistic regression was used to determine if nest site abundance was a good predictor of artificial net survival. No relationship between nest site abundance and survival of artificial nests was found on either control (X^2 = 3.81, df = 1, P = 0.051) nor treatment sites (X^2 = 0.047, df = 1, P = 0.829) thus suggesting that nest site abundance is not a good predictor of artificial nest survival.

2001 - Average percent of artificial nests that survived 28 days was higher on control versus treatment sites (control [n = 16, $\bar{x}$ = 84.37, SD = 12.85], treatment [n = 16, $\bar{x}$ = 40.63, SD = 31.67]). Trapping of specific mesomammals had a significant effect on survival of artificial nests; however, trapping did not cause an increase in the survival of artificial nests (F = 29.078, df = 1, P = 0.000). Greater nest site availability (i.e. suitable nest sites/ha) on control ($\bar{x}$ = 554, SE = 106) versus treatment areas ($\bar{x}$ = 236, SE = 47) suggested a possible relationship between artificial nest survival and nest site availability. No relationship between survival of artificial nests and number of suitable nest sites was indicated (F = 1.999, df = 1, P = 0.168, R^2 = 0.062). Logistic regression was used to determine if nest site abundance was a good predictor of artificial net survival. No relationship between nest site abundance and survival of artificial nests was found on either control (X^2 = 0.374, df = 1, P = 0.541) nor treatment sites (X^2 = 0.324, df = 1, P = 0.569) suggesting nest site abundance is not a good predictor of artificial nest survival.

Discussion

Predator control does not appear to be a viable method of increasing survival of nests when applied to small areas over short periods of time. Landowner constraints on trapping methods affect the feasibility of predator removal as a mechanism to increase survival of nests. Small-scale trapping efforts also are affected by the behavior of the animal (i.e., trap happy, trap shy). In portions of these studies, the inability to remove skunks was exacerbated by the inability to adjust trapping methods. Landowners lacking the means and area necessary for large-scale-predator control should not rely on predator removal as a mechanism for increasing survival of nests.

In western Texas, medium-sized carnivores, or mesomammals, such as foxes (*Urocyon cinereoargenteus* and *Vulpes vulpes*), raccoons, feral cats, and skunks are primary nest predators (Dolbeer et al. 1996, Hernandez et al. 1997, Slater et al. 2001). Removal of these predators may cause non-target species such as coyotes to switch prey items and focus primarily on nests. Prey switching may explain the higher artificial nest survival on control versus treatment sites in 3 of 4 years.

Fragmentation of habitats may exacerbate nest depredation. Small fragmented areas may allow for concentration of predators, which would increase their likelihood of encountering a nest. Artificial nests in prairies <15 ha were depredated more (37.0%) than those in larger prairies (13.9%) during a study in southwestern Missouri (Burger et al. 1994). Once a nest had been located, it is possible that predators focused their search efforts in the general vicinity. This effect may be compounded by habitat fragmentation. Placing nests along transect lines may also have aided predators in locating nests, thus increasing nest depredation. However, we assumed that any bias due to nest placement or nest density was spread evenly across all sites.

It also has been hypothesized that predators follow human scent to aid in the location of nests (Bayne and Hobson 1997, Sloan et al. 1998). Skagen et al. (1999) reported that nest predation was not higher when human scent was present. Donalty and Henke (2001) demonstrated that attempts to conceal scent were ineffective. We also assumed that any bias due to human scent left near nests was spread evenly across all sites.

Slater et al. (2001) suggested a threshold of nesting sites (≥754 nest sites / ha) at which the effects of nest depredation were mitigated. Although there was 1 site where a statistically significant relationship between simulated nest survival and nest site availability was indicated (e.g., 1999 $P = 0.010$), this result does not appear to be biologically significant. Therefore, nest site abundance does not appear to be a good predictor of artificial nest survival. Although nest site availability numbers were higher on all sites than the threshold established by Slater et al. (2001), intensive year-round grazing, and below average rainfall for several years may have contributed to high nest depredation. Intensive grazing during drought periods prior to the studies being initiated resulted in nests placed in the already fragmented bunchgrass area likely becoming even more vulnerable to depredation. Even though grazing pressure was reduced on the Coleman County area during the study, the intensity of the drought may not have allowed for the reparation of nesting habitat.

Management of brush and nesting habitat may be the most important factor in quail ecology. Proper amounts of both can increase nest success and survival. Drought and overgrazing decrease the amount of nesting cover, thus decreasing nest survival. High numbers of nest sites may not provide all the protection quail need to decrease nest depredation. For instance, nest success was greater in Kansas when quail were able to nest in tall clumps of grass (Taylor et al. 1999). This suggests that quality may be as important as quantity of nest sites.

Overall, a partial removal of mesomammal predators over a short period of time did not appear to affect the predator population, nor increase survival of artificial nests. Given that some nest predators were not targeted in these studies, we are uncertain of the effectiveness of predator removal. Given the result of our data, we suggest that a 1-month-trapping period is insufficient in areas where bobwhites nest.

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Survival of Female Scaled Quail During the Breeding Season at Three Sites in the Chihuahuan Desert

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Scaled quail (*Callipepla squamata*) populations declined markedly across much of their range from 1988-2004, however little research has been conducted to investigate possible causes for the decline. As part of a larger study on scaled quail ecology and management, and in an attempt to determine whether breeding season survival could be implicated in this decline, we monitored survival of radiotagged female scaled quail during the breeding season at sites in Brewster and Pecos counties, Texas, and Sierra County, New Mexico, USA during 1999-2003. Survival rates were calculated using Kaplan Meier analysis for birds living $>$7 days post capture. Interval survival rates (*S*) from Mar-Aug ranged from 0.46 to 0.82 for populations in Pecos County, Texas, and from 0.56 to 0.69 in Brewster County, Texas; survival was lower at the New Mexico site (*S* ranging from 0.22 to 0.48). Predation by mammals was the leading cause of mortality at both Texas study sites, whereas predation by raptors was the primary cause of mortality at the New Mexico site. Several mortalities in Texas were attributed to drowning; 3 in a water trough and 2 others following a flash flood. Survival rates on Texas sites were not affected by moist soil management but were greater than survival on New Mexico sites. Survival on New Mexico sites was greater on areas with access to free-choice quail feeders (*S* = 0.48) relative to a non-fed site (*S* = 0.22). Survival rates of scaled quail during the breeding season were higher than those reported for radiotagged northern bobwhite in west Texas at similar latitudes. Survival of female scaled quail during the breeding season does not appear to be a bottleneck to recruitment, at least not on sites where conservative grazing management is practiced.

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Key words: *Callipepla squamata*, Chihuahuan Desert, management, New Mexico, predation, radiotelemetry, scaled quail, survival, Texas

Introduction

Abundance of scaled quail (*Callipepla squamata*) declined for undocumented reasons throughout most of their range from 1966-2004 (Schemnitz 1994, Rollins 2000, Sauer et al. 2005) (Figure 1). Predators (Rollins 2000), drought (Wallmo and Uzzell 1958, Saiwana et al. 1998, Pleasant et al. 2006), disease (Rollins 2000), overgrazing (Ligon 1937), changing habitat conditions (Schemnitz 1994, Rollins 2000), or some combination of these factors (Bridges et al. 2001) have been cited as possible mechanisms for declining trends in scaled quail in the Chihuahuan Desert.

Studies of scaled quail have lagged notably behind those of northern bobwhite (*Colinus virginianus*) especially since the advent of radio telemetry (Rollins 2000). Earlier studies by Bent (1932), Wallmo (1956*b*), Schemnitz (1961), and Campbell et al. (1973) were based on field observations and provided general ecological information about scaled quail, but provided little information on nesting ecology (because of the difficulty of locating nests), movements, or population dynamics (e.g., cause-specific mortality). Recently Pleasant et al.

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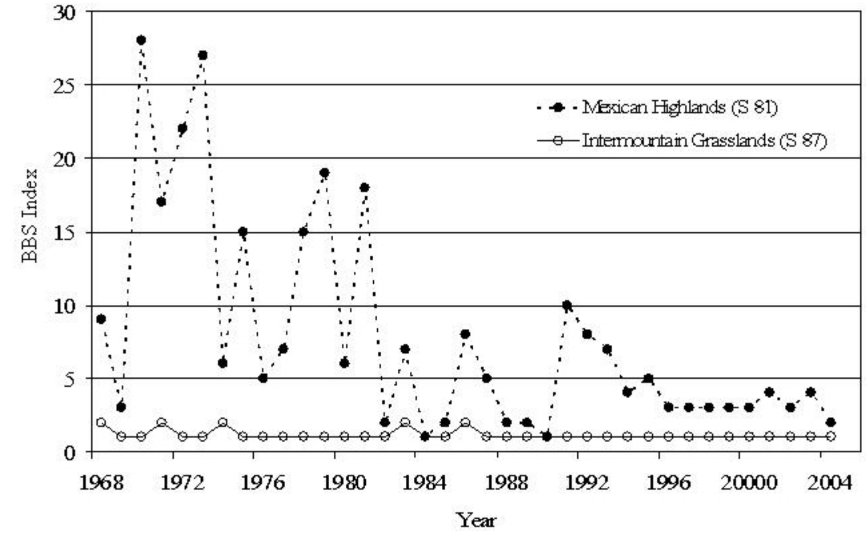

Figure 1: Scaled quail trends from the Breeding Bird Survey for the Mexican Highlands (Sites 1 and 2 in this study) and Intermountain Grasslands (Site 3), 1968–2005 (Sauer et al. 2005).

(2006) used radiotelemetry to address information voids like nesting ecology and survival. We report survival and cause-specific mortality for 3 populations of radiotagged scaled quail during the breeding season in west Texas and south-central New Mexico. We also report impacts of moist-soil management and supplemental feeding on summer survival of scaled quail.

Study Area

Three study sites were involved: 2 in Texas (Pecos and Brewster counties) and 1 in New Mexico (Sierra County) (Figure 2). Site 1 was the Sherman Hammond Ranch, a 12,000-ha private ranch located about 40 km southwest of Ft. Stockton, Pecos County, Texas. Site 2 was the 18,000-ha Texas Parks and Wildlife Department's Elephant Mountain Wildlife Management Area located 65 km south of Alpine, Brewster County, Texas. Site 3 was the Armendaris Ranch, a 120,000-ha private ranch located 20 km east of Truth or Consequences, Sierra County, New Mexico. Sites 1 and 2 are located in the Mexican Highlands Bird Conservation Region (BCR) and Site 3 is located in the Intermountain Grasslands BCR.

Sites 1 and 2 were used to examine differences in areas with or without moist-soil areas created by water harvesting techniques (i.e., spreader dams). These experiments were conducted in 1999 and 2000 (Site 1) and 2000 and 2001 (Site 2). Site 1 had 3 study populations each separated by >12 km: (1) a "Negative Control", i.e., an area characterized by the absence of spreader dams; (2) a "Positive Control", i.e., an area surrounding the ranch headquarters with a 2-ha irrigated lawn (thus providing scaled quail access to green vegetation), and (3) a "Treatment" area characterized by a landscape punctuated with moist-soil sites following rainfall events. Site 2 was similar to Site 1 except no positive control was available and the overall number of moist-soil sites was considerably lower.

Vegetation at Site 1 was dominated by desert scrub and consisted mainly of creosote (*Larrea tridentata*), tarbush (*Flourensia cernua*), and honey mesquite (*Prosopis glandulosa*). Common grasses include tobosa (*Pleuraphis belangeri*) and bush muhly (*Muhlenbergia porteri*). The average annual precipitation in Ft. Stockton is 305 mm, with most of it falling between May and October. The average temperature is 8° C during winter and 27° C in the summer.

Major plant communities at Site 2 vary from

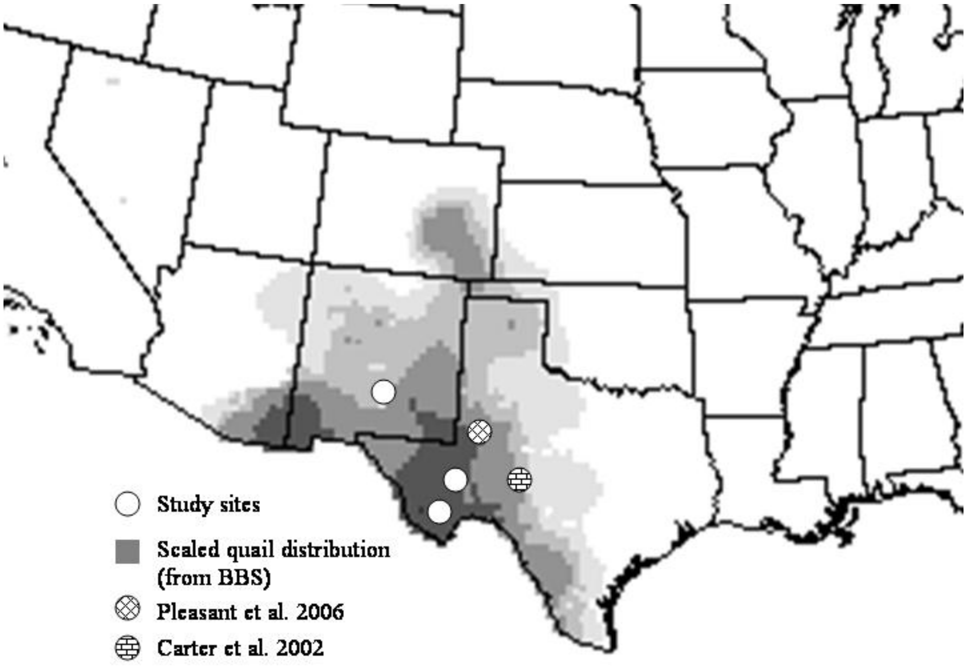

Figure 2: Study sites (open circles) relative to range of scaled quail (Sauer et al. 2005) and recent studies on scaled quail in west Texas.

grama (*Bouteloua* spp.) and tobosa- dominated grasslands to Chihuahuan Desert scrubland. Upland areas contain mesquite and redberry juniper (*Juniperus pinchotii*). Average annual precipitation at Elephant Mountain WMA from 1986 to 2001 was 363 mm. The driest year during this period was 2001, when only 205 mm of precipitation were recorded. Additional details on Sites 1 and 2 are reported by Buntyn (2004) and Lerich (2002), respectively.

Site 3 was used to evaluate differences in survival between areas with or without year-round supplemental feeding with milo. We monitored scaled quail on 2 areas; a treatment area that featured free-choice quail feeders, and a nonfed control. This experiment was conducted from Oct 2002 -August 2003. The control area was located 7 km north of the fed area, a distance well beyond what scaled quail normally travel, and included feeders that had been active for > 4 years prior to our study. Scaled quail at both sites had access to water via guzzler devices (Rollins et al. 2006). Access to water was not restricted in this study; hence, the presence of guzzlers (i.e., water) was not a treatment variable. Spacing between quail feeders in both the treatment and control areas was approximately 1 feeder per 1.1 km of road. Feeders in the treatment area were constructed of 206-l plastic barrels with 10-12 small (1-cm) holes placed at intervals ranging from 7-25 cm from the bottom of the barrel. These feeders are free choice as milo was available at any time throughout the day and year-round. The 5 feeders in the control were timed feeders, which prior to them being turned off in October 2001 (prior to onset of this study), were on timers set to disburse milo on the ground directly beneath the feeder for 3 seconds at 7 a.m. and 4 p.m. It should be noted that each feeder site (active or not) was a site for trapping, and although feeders were shut off in the control area, approximately a cup of milo remained as bait for each day of trapping. Therefore, a limited amount of milo was available at trap sites in the control area for trapping purposes. Strictly speaking, the control area might be better considered a 'limited' feed area and the treatment an ad libitum feed area.

The portion of Site 3 used for this study was a 2,500-ha section near the southern border of the ranch. It was predominantly black grama (*B. eriopoda*) grassland in good to excellent range condition. Mesquite was the major woody plant and yuccas (*Yucca* spp.) were primary succulents. Annual precipitation at this site averages 270 mm. The study area was mired in drought for the duration of the study–annual precipitation in 2001 and 2002 was only 53 and 44% of the long-term means, respectively (Western Regional Climate Center 2003). Above normal temperatures prevailed during this time; for example, June 2002 had the highest mean monthly temperature on record.

Methods

Trapping and telemetry

Scaled quail were trapped in funnel traps on each study site during February-April. The study periods were 1999-2000 (Site 1), 2000-2001 (Site 2), and 2002 (Site 3). Female scaled quail were fitted with neck-loop telemeters weighing <7 g and equipped with mortality sensors. Telemeters from 2 different manufacturers were used; Telemetry Solutions (Concord, CA) was used for Sites 1 and 2, while those manufactured by Wildlife Materials, Inc. (Carbondale, Illinois) were used for Site 3. All quail captured were leg-banded with individually-numbered aluminum bands. Quail were aged and sexed according to methods described by Wallmo (1956*a*). Trapping and handling methods were approved by the Texas A&M University's Care of Laboratory Animal Welfare Care and Use Committee.

Monitoring

Radiotagged birds were tracked via homing 2-3 times per week using 3-element Yagi antennas with portable receivers. Radiotagged quail were monitored <3 times a week from time of capture (mid-February-March) until time of death, or censoring from the study, through the breeding season (i.e., August) during each year. At the onset of incubation, quail were monitored daily, while attempting to not disturb the nesting hen. All mortality signals were investigated upon detection and cause of death was determined using criteria described by Carter et al. (2002). Cause-specific mortality was reported as avian, mammalian, snake, exposure, or unknown.

Data analyses

Interval survival was calculated using Kaplan and Meier (1958) analysis for birds living >7 days post-capture. Staggered entries of individual quail were analyzed as described in Pollock et al. (1989). Individuals lost to emigration or radio failure, were censored (Pollock et al. 1989). A log-rank Chi-square test was used to evaluate differences in survival between treatments (or years) at a particular site. Assumptions included: (a) censorship of individuals was random and survival times were independent for all radiotagged birds; (b) survival of quail was not affected by capturing, handling, or radiotagging; (c) survival times were independent; (d) radiotagging did not influence future survival; (e) censoring mechanisms were random; and (f) newly radiotagged quail had the same survival function as previously radiotagged quail (Pollock et al. 1989).

Results

Site 1 - Pecos County, Texas

We trapped a total of 269 birds in 1999 and 228 birds in 2000. The 1999 sample consisted of 154 females (57%) and 115 males (43%); most (72%; n = 193) were adults. Of these, 120 females (75% adults) were radiotagged (40 per study site). The 2000 sample included 136 females (59%); 42 (11%) were adults and 186 (89%) were subadults. A total of 90 females were radiotagged (30 per study area). The population trapped in 2000 was consistently younger across all study areas, with adults comprising only 16% of the sample across study areas.

Survival across all sites was similar between years (Figure 3) and across treatments in 1999 (Table 1). In 2000, survival in the Positive Control (S = 0.47±0.38) was lower than that in either the Treatment (S = 0.81±0.02) or Negative Control areas (S = 0.82±0.03) ($\chi^2 = 14.3$, $v = 1$, $P < 0.005$).

A total of 20 mortalities were observed in 1999

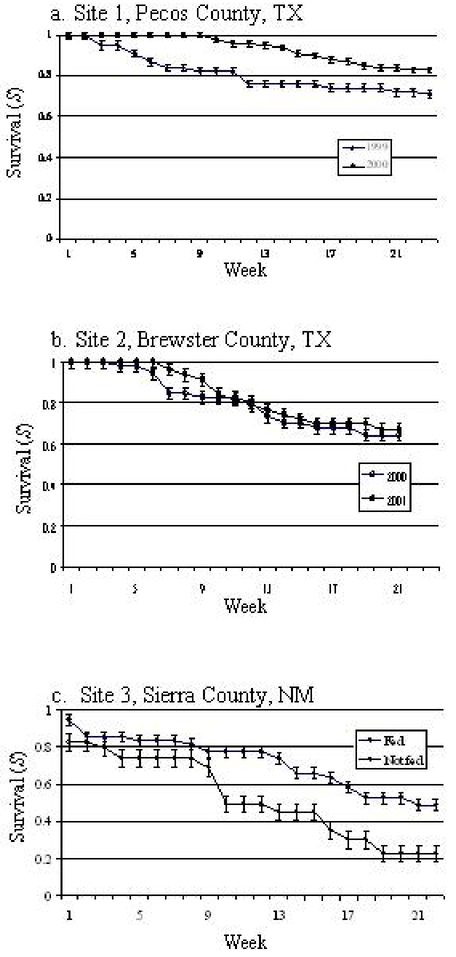

Figure 3: Survival of female scaled quail during spring and summer at 3 sites in the Chihuhuan Desert, 1999–2002.

(Figure 4); most (n = 14) were attributed to mammalian predators while raptors accounted for 4 kills. The remaining 2 mortalities were caused by exposure (flooding). Predation by mammals was also the leading agent of mortality in 2000, accounting for 18 of 23 mortalities; raptors accounted for 4 kills and 1 bird was killed by a western diamondback rattlesnake (*Crotalus atrox*). Predation was similar between years and among sites with the exception of the Positive Control in 2000, where 6 mortalities were attributed to free-ranging cats. All 6 mortalities were recorded within 500 m of the ranch headquarters and their telemeters were retrieved from areas frequented by cats. We estimated the free-ranging cat population to consist of 5 individuals in 1999 and 13 individuals in 2000.

The 2 mortalities related to exposure in 1999 were observed following a flash flooding event from a heavy rain that occurred on 17 June 1999 about 6 km up the watershed. Rainfall totaling15 cm fell during a short period causing a sheet of water to blanket the Positive Control site. The following day, 2 mortalities were recorded-we attributed both to the flash flood event. Each bird's carcass was located under a layer of silt (>10 cm). Each of the hens involved had a brood of chicks (<10 days of age) at the time.

Table 1: Breeding season (March−August) estimates of survival for radio-tagged female scaled quail at various sites in the Chihuahuan Desert, 1999−2002, including estimates from other recent studies.

Site	Years	*N*	*S*	*SE*	Reference
Irion Co., Texas	1994	17	0.7	*Not reported*	Rollins 2000
Bailey Co., Texas	1999	66	0.38	*Not reported*	Pleasant et al. 2006
Bailey Co., Texas	2000	72	0.30-0.43	*Not reported*	Pleasant et al. 2006
Pecos Co., Texas	1999	120	0.8	0.03	This study
Pecos Co., Texas	2000	90	0.71	0.08	This study
Brewster Co., Texas	2000	62	0.63	0.05	This study
Brewster Co., Texas	2001	46	0.67	0.04	This study
Sierra Co., New Mexico, fed site	2002	132	0.48	0.08	This study
Sierra Co., New Mexico, not fed	2002	77	0.22	0.08	This study

Site 2 - Brewster County, Texas

We radiotagged a total of 72 birds in 2000 (70 females, 2 males) and 46 birds (25 females, 21 males) in 2001. Seasonal survival was similar between years and between sites in both 2000 (S = 0.69±0.09 and 0.56±0.10) and 2001 (S = 0.67±0.10 for both sites) (Figure 3). Cause-specific mortality for 2000 was attributed to mammalian predators (n = 7 kills), unknown predators (n = 6 kills), raptors (n = 2 kills), and drowning (n = 3 deaths) (Figure 4).

Site 3 - Sierra County, New Mexico

We trapped a total 532 scaled quail; the sex ratio of birds trapped in 2002 was 1.04:1 females:males (271 females and 261 males). The age ratio of this sample was 2.86:1 subadults:adults (i.e., 74% subadults). A total of 209 hens were radiotagged: 132 (46 adults, 86 subadults) in the treatment area and 77 (20 adults, 57 subadults) in the control area. Survival on site 3 was lower than on sites 1 and 2 but was higher in the fed area (S = 0.48±0.08) than the area not fed (S = 0.22±0.08; χ^2 = 4.57, 1 v = 1, P = 0.03) (Figure 3). The 2 populations survived similarly until late-April when hens in the control portion of the ranch began suffering greater mortality.

A total of 88 mortalities of radiotagged quail were observed; 34% (n = 30 kills) were attributed to mammals, 53% (n = 47 kills) to raptors, and 7% (n = 6 kills) were characterized as unknown (Figure 4). Five mortalities (6%) were attributed to handling, i.e., where the telemeters were initially fitted too tightly. Cause-specific mortality was similar between fed and non-fed sites. Aside from a kit fox (*Vulpes macrotis*) that killed 3 juvenile cocks in a trap, we did not observe any mortalities at or immediately adjacent to quail feeders.

We experienced chronic problems with battery failure in the telemeters used (both brands) at Sites 2 (Telemetry Solutions) and Site 3 (Wildlife Materials, Inc). The batteries seldom, if ever, lasted as long as described by the manufacturer (270 days). Typically, collars lasted <120 days making it difficult to obtain long-term data series for specific females.

Discussion

Data from these telemetry studies suggest survival rates of female scaled quail during spring and summer were quite high, especially at the 2 sites in Texas. Rollins (2000) documented survival of a small sample (n = 17) of scaled quail at 0.70 from January-August in Irion County, Texas, in 1995 (about 225 km east of Site 1). Pleasant et al. (2006) reported survival of female scaled quail in Bailey County, Texas (about 200 km north of Site 1) during the same time period of our study (1999-2000) ranging from 0.30-0.43.

Survival rates we observed, especially at the

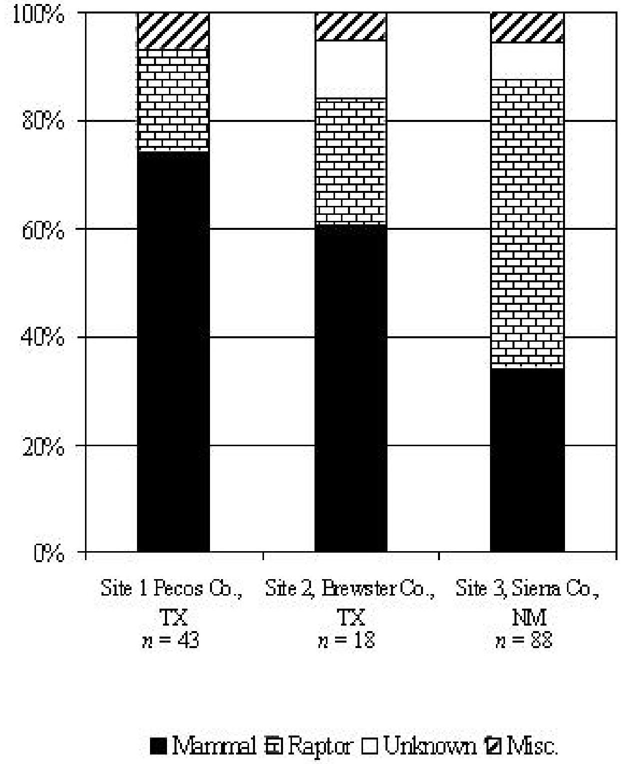

Figure 4: Cause-specific mortality of scaled quail during spring and summer at 3 sites in the Chihuahuan Desert, 1999−2002.

Texas sites, were generally greater than observed for female bobwhites during the breeding season across most of their range. Carter et al. (2002) reported relatively low survival ($S = 0.17$) of 54 radiotagged bobwhites during the breeding season in Irion County, Texas (about 250 km east of Site 1) compared to 0.70 for scaled quail on the same site. Hernandez et al. (2003) reported summer survival rates from 0.17-0.61 in Shackelford County, Texas (about 400 km east of Site 1). Brooks (2005) reported breeding season survival rates of 0.50 for female bobwhites in Fisher County, Texas (about 250 km east of Site 1). Greater survival rates of scaled quail (relative to bobwhites) may be related to less abundant predator populations in more arid environments, or inherent differences between the vulnerability of bobwhite and scaled quail relative to predators (Rollins and Carroll 2001) or hunting (Rollins 2000). Lehmann (1984, :225) considered scaled quail to be more intelligent than bobwhites.

Mammalian predators were the primary cause of mortality for female scaled quail during spring and summer at the 2 sites in Texas, but raptors caused proportionally more mortalities (about twice as many) in New Mexico. Miscellaneous sources of mortality included drowning, rattlesnakes, exposure (hailstorm), and drowning. Hernandez (1999), Carter et al. (2002), and Brooks (2005) reported similar cause-specific mortality sources for female bobwhites during spring and summer in west Texas. Red-tailed (*Buteo jamaicensis*), and Swainson's hawks (*B. swainsoni*) were the 2 most common species of raptors observed at the New Mexico site; neither are regarded as particularly efficient predators of quail. Great horned owls (*Bubo virginianus*) are common at all sites, and have been known to prey on scaled quail during the breeding season (Carter et al. 2002). Cooper's hawks (*Accipiter cooperi*) and northern harriers (*Circus cyaneus*), are present through April in this area, and were likely responsible for most of the raptor-caused mortalities during the spring. Goodwin and Hungerford (1977) indicated that most scaled quail kills in Arizona were made by avian predators including northern harrier, red-tailed hawk, prairie falcon (*Falco mexicanus*), and great-horned owl. Campbell et al. (1973) indicated common scaled quail predators in New Mexico included hawks, owls, coyotes (*Canis latrans*), and snakes.

Scent station indices indicated low diversity and prevalence of mammalian predators at all 3 sites (Lerich 2002, Buntyn 2004, , T. D. Sparks, unpublished data). Low abundance of meso-mammals in this portion of the Chihuhuan desert may have permitted greater survival and hatch rates of scaled quail (Buntyn 2004). The impact of free-ranging cats on scaled quail underscores the potential significance of feral and free-ranging cats on quail.

Rollins and Carroll (2001) discussed impacts predators may have on scaled quail. Sauer et al. (2005) documented a >2.0% increase per year in accipiter abundance over a large area of the U. S. since 1967. Cooper's hawks, considered to be the most efficient predator of bobwhites, are present throughout the range of scaled quail at least through spring. Northern harriers are common winter residents (through early April) and have been identified as accomplished predators of quail (bobwhite and scaled quail) (Jackson 1947). The only practical approach to minimize raptor losses of scaled quail is to provide adequate loafing and screening cover (Rollins and Carroll 2001). Conservative grazing management was practiced at each of our study sites, which coupled with the presence of suitable screening cover (e.g., catclaw mimosa), likely afforded scaled quail greater survival than which might be expected over much of the Chihuahuan Desert.

The drownings of 2 hens with broods following a flash flood event at Site 1 are intriguing. We speculate the hens' maternal instincts to brood their young chicks in the presence of rising water precluded their escape. We can offer no other reason why adult birds would succumb to rising flood waters.

Three radiotagged hens drowned in the same water trough at Site 2; 2 in 2000 and 1 in 2001. The trough is <1 m high and approximately 2 m in diameter. Many other bird species used this trough but only scaled quail were found drowned. The water level in the trough was always <3 cm from the rim. It is unclear whether the telemeters had anything to do with their drowning (no other radio-marked quail were found drowned), or if some site (trough) specific factor played a role in the drownings. There are >30 water troughs of the same style spread out on Elephant Mountain WMA and no other quail had ever been found drowned in any of them. These drowning incidents underscore idea by Schemnitz et al. (1998) of making such troughs escapable by quail.

Management impacts on survival

Moist-soil management

We initiated our investigations into breeding season dynamics of female scaled quail in 1999 in an attempt to explain why Site 1 had maintained a viable, huntable population during a time period (1990s) when scaled quail abundance had declined markedly across much of their range in Texas (Rollins 2000). During the 1990s the Ranch appeared to have greater quail abundance than those of surrounding ranches. A landscape punctuated by spreader dams was hypothesized to be a major component in the population abundance. The distribution and abundance of spreader dams appeared to offer additional habitat components i.e., cover and possibly additional food resources (arthropods and seeds). Indeed, herbage and arthropod biomass were 25 and 6 times greater, respectively, on moist-soil sites relative to adjacent uplands (Buntyn 2004). However, the presence of spreader dams did not affect scaled quail survival during the breeding season at either Site 1 or 2 in either years when precipitation was above (e.g., 1999) or below normal (e.g., 2000). During the study period (April-August) spreader dams held water for short periods, yet evaporation eliminated standing water within a couple of days. The availability of free-standing water (i.e., for drinking) has not been shown to increase scaled quail abundance (Wallmo and Uzzell 1958). We recognize that benefits of moist soil management (e.g., increased seed or arthropod production (Buntyn 2004)) may accrue to scaled quail and possibly increase either brood or fall-winter survival, but we did not measure these parameters.

Grazing management

Livestock grazing and its relationship to scaled quail habitat need additional study (Saiwana et al. 1998, Smith et al. 1996, Rollins 2007). Smith (1996) found no scaled quail present on an ungrazed site in the northern Chihuahuan Desert, suggesting that heavy or thick vegetation is not frequented by scaled quail. Scaled quail prefer areas of open ground yet require cover to avoid predation from both mammalian and avian predators. In south Texas, an area of sympatry with bobwhites, scaled quail used areas of sparse vegetation compared to areas of dense herbaceous cover used by northern bobwhites (Wilson and Crawford 1987). Conversely, Rollins (2007) suggested "undergrazing", i.e., stocking rates perhaps 30-50% below those historically practiced for improving overgrazed habitats for bobwhite and scaled quail in west Texas. Grazing management is especially important during times of drought conditions such as those that characterize the Chihuahuan desert (Campbell-Kissock et al. 1984, Nelson et al. 1997).

Supplemental feeding

We observed greater survival of female scaled quail during one breeding season at a fed site compared to a control site however both sites had survival rates lower than the Texas sites we studied. Providing supplemental feed to quail has typically been dismissed by quail biologists as either ineffective (Guthery 2002, :149), inefficient (Guthery et al. 2004), or too expensive (Campbell 1959). However, some private landowners may have the capital to accommodate supplementation and can control some other factors (e.g., hunting pressure) that cannot be controlled on public land. Rollins (2000) reported frequent visitations of adults and young chicks (<3 weeks old) to feeders in west Texas, and recommended that supplemental feeding be evaluated as a management tool in west Texas. Rollins et al. (2006) recorded scaled quail and their broods using feeders commonly at this study site (19 to 22% of feeder visitations), or about twice as often as bobwhites at 4 sites in west Texas (6 to 10% of feeder visitations; Henson 2006). Feeders are also effective in making quail hunting more predictable and productive (Rollins 2000).

Year-round supplementation with milo, in addition to providing a formulated egg-laying ration, significantly improved survival and reproduction of bobwhites on fed sites in northern Florida (Sisson et al. 2000, Tall Timbers Research Station 2001). Benefits from providing supplemental feed might be more important for scaled quail under weather conditions similar to those we encountered during our study (below normal precipitation and above normal temperatures). Additional research is needed to better elucidate the potential benefits of supplemental feeding for desert quails.

We acknowledge some limitations within our studies. The feeder study on Site 3 was non-replicated, and Sites 1 and 2 were not replications in the strictest sense as no positive control at Site 2 and the number of spreader dams was perhaps 10% of that found on Site 1. The feeder study was conducted during a period of severe drought; whether we would have observed treatment differences in more moderate weather conditions is unknown. Our study sites, especially Sites 1 and 3, were atypical for their respective regions because of their conservative grazing management and subsequent higher seral stages.

Acknowledgments

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Survival and Causes of Mortality for Northern Bobwhites in the Southeastern USA

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Long-term studies are imperative to increase our knowledge of northern bobwhite (*Colinus virginianus*; hereafter bobwhite) demographics. During 1992-2005, we determined survival and cause-specific mortality of bobwhites on 10 study areas in southern Georgia and eastern Alabama, USA. We radio-tagged 7,105 bobwhites and determined 49 annual (Oct-Sep) and 110 seasonal survival estimates to examine spatial and temporal variation in survival. Annual survival for all sites and years combined averaged 0.196 (SE = 0.011) and ranged from 0.08 to 0.40. Over-winter (Oct - Mar) seasonal survival estimates (n = 51) averaged 0.541 (SE = 0.019) and ranged between 0.25 and 0.82, while breeding season (Apr-Sep) survival estimates (n = 59) averaged 0.352 (SE= 0.013) and ranged between 0.13 and 0.59. Over-winter mortality (n = 1,473) of known fates was attributed to avian predation (0.572 ± 0.040), mammal predation (0.265 ± 0.044), harvest (0.156 ± 0.028), snake predation (0.001 ± 0.004) and other (0.005 ± 0.002). Breeding season mortality (n = 2138) was attributed to avian predation (0.613 ± 0.026), mammal predation (0.339 ± 0.049), snake predation (0.037 ± 0.006) and other (0.011 ± 0.004). These over-winter survival estimates were higher than previously published estimates for populations on unmanaged lands and/or heavily harvested populations. On managed lands in the Southeast, bobwhite annual survival rates derived from radio-telemetry were reasonable and provided useful information for management and research.

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Key words: *Colinus virginianus*, mortality, northern bobwhite, southeastern USA, survival, telemetry

Introduction

Understanding survival rates and causes of mortality for northern bobwhite (*Colinus virginianus*) populations is critical for making site specific management decisions as well as for regional recovery efforts. Cox et al. (2004) stated that "the need remains for site-specific, descriptive data on bobwhite survival and mortality because these basic descriptive data are prerequisite for development of sound management strategies." The need for reliable, long-term demographic information on bobwhites is heightened by their long-term decline throughout most of the Southeast. A number of studies over the years have provided some survival information for bobwhites in the Southeast (Guthery and Lusk 2004); however, the majority of these studies were short in duration and based on small sample sizes (Terhune et al. 2007). In addition, the reliability of these telemetry based survival estimates has recently been called into question (Guthery and Lusk 2004) insinuating that all survival estimates derived with radio-telemetry are biased low. Therefore, it is important to present survival estimates for bobwhite populations in the Southeast from longer term studies derived from those with relatively large sample sizes.

In an effort to produce this much needed long-term data, Auburn University's School of Forestry and Wildlife Sciences began a long-term investigation of bobwhite population ecology, management,

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and hunting on private plantations in South Georgia and Alabama in 1992 commonly known as the Albany Quail Project (AQP). One objective of this long-term radio-telemetry project was to collect reliable data on survival and causes of mortality on properties that were being actively managed for bobwhites. The objectives of this paper are to: (1) present annual and seasonal survival estimates for 10 properties over the first 14 years of the AQP (1992 - 2005); (2) examine the annual variation and long-term means of bobwhite survival both within and among populations; (3) present cause specific mortality estimates for these populations; (4) compare all these estimates to previously published estimates; and (5) determine if our long term survival estimates are biologically reasonable given bobwhite populations on our study areas are stable or increasing.

Study Area

These data were collected on 13 separate study sites on 10 large private landholdings in southern Georgia and eastern Alabama (Figure 1) that combined encompass approximately 42,000 ha. All but one of these (site 5) had active and intensive quail management programs in place. Five of these properties (sites 1-5) were contiguous and centered around our headquarters south of Albany, GA in Baker County. Three others (sites 6-8) were scattered around southern GA, and the remaining 2 (9, 10) were in east central Alabama. Following is a brief description of each:

Site 1

The primary study site and headquarters for AQP was a 6,000 ha private plantation in Baker county, GA with a history of quail management dating back to the 1940's. Typical of the properties in the region, it was characterized by mature old-field pine forests (80%) with a low basal area (3-9 m^2/ha) and scattered 1-4 ha fallow fields (20%). Management techniques included maintenance of an open canopy through pine timber thinning and hardwood midstory removal, frequent prescribed burning in the woods, and seasonal disking of fields to stimulate weeds and insects, drum-chopping and mowing, supplemental feeding, and mammalian nest predator control. As a result of this management program, quail abundance estimates in recent years averaged approximately 5 birds/ha. More detailed descriptions of this property can be found in (Yates et al. 1995, Burger et al. 1998, Hughes et al. 2005, Terhune et al. 2006, 2007).

Site 2

The secondary study area for AQP, this 4,400 ha plantation straddles the border of Dougherty and Baker Counties, GA and has a history, management program, and quail abundance estimates very similar to site 1. More detailed descriptions of this property can be found in Simpson (1976), Sisson et al. (2000*b*,*a*) and Terhune et al. (2006, 2007).

Site 3

This 4,800 ha plantation in Baker County, GA also has a similar history, management program, and quail abundance estimates as site 1 and 2. This property was divided into 2 study sites (3a, 3b) for a large scale nest predator management experiment in conjunction with United States Department of Agriculture - Wildlife Services, Tall Timbers Research Station, and the University of Georgia. 3a was a 2,000 ha block on the east side of the property and 3b was a 2,000 ha block on the west side. These sites were alternately used as treatment and control sites with data collected and analyzed separately.

Site 4

This 2,800 ha plantation in Dougherty County, GA is within the matrix of plantations south of Albany and also has a history, management program, and bird density very similar to the first 3.

Site 5

This 2,000 ha property was a large private farm with no active quail management program. Located in Baker County, GA this property was dominated by center-pivot irrigated row crop fields (65%) with the remainder a mixture of young planted pines and a large creek swamp. A more detailed description can be found in Hughes et al. (2005) and Terhune et al. (2006).

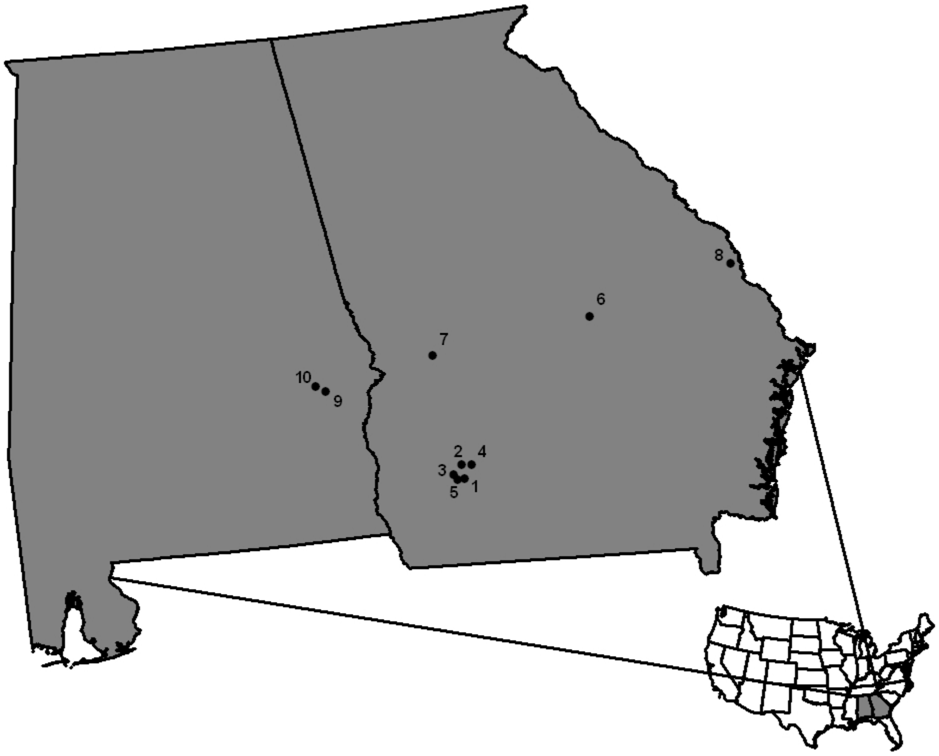

Figure 1: Location of 10 study areas sites of the Albany Quail Project in Georgia and Alabama during 1992-2005.

Site 6

This 3,200 acre property was located in Laurens and Bleckley Counties, GA and was a working row crop farm that was intensively modified to improve it for bobwhite production and hunting. Although in the upper coastal plain, this area was not known for bobwhite abundance therefore it was one of the first in the area to manage specifically for quail. The property was a mixture of agricultural fields, open canopy woodlands, hedgerows, and young planted longleaf. Bobwhite density at the time of the study approached 2 birds/ha. More detailed descriptions of this site can be found in Sisson et al. (2002*b*) and Terhune et al. (2009).

Site 7

This 1,200 ha property was located in Marion and Schley Counties, GA in the transition area between the Coastal Plain and Piedmont commonly known as the Fall Line. This area was well outside the traditional quail belt in Georgia but was intensively modified by a new owner to create high quality quail habitat. Cover types were a mixture of open canopy woodlands, young planted longleaf, and fallow fields. This was the study site for our wild quail relocation study thereby generating 2 sets of data: 7a) relocated wild birds and 7b) resident wild birds.

Site 8

This 9,200 acre plantation was in southeast Georgia in Screven and Burke County. It has a long history of wild quail management similar to the properties in the Albany area with the main difference being less mature pine woods and more open land, including row crop land, interspersed throughout the quail courses. Bobwhite densities on this property are approximately 2 birds/ha. There were 2 study sites on this property as well. 8a was a mostly open landscape with a high percentage of the ground under center pivot irrigation while 8b was a more wooded site with no irrigation.

Site 9

This 5,200 acre plantation was located in Russell and Barbour County, Alabama near the town of Hurtsboro. This property was mostly gently rolling pine woodlands interspersed with small fallow fields that made up about 10% of the landscape. History, management, and bird density was very similar to the Albany plantations. The biggest difference was the wet nature and low pH of the soils.

Site 10

This 2,620 ha property is contiguous with site 9 and has a similar management program. The land is somewhat different as it is more of a prairie soil with a very diverse groundcover and more of the land is open. There are less fallow fields as well and the bird density here is about 2 birds/ha.

Methods

All sites were part of the on-going work of the Albany Quail Project with research protocols similar among sites. Wild bobwhites were trapped on active study sites twice each year (Mar-Apr and Oct-Nov) during 1992-2005 using standard, baited funnel traps (Stoddard 1931). Each bird was classified by age and sex, weighed, leg banded, radio-tagged and released at the capture site. Only birds weighing ≥132g (<5% of body weight) were outfitted with pendant style transmitters (6.0g) equipped with an activity switch (Holohil Systems, Ltd., Ontario, Canada). Trapping, handling, and marking procedures were consistent with the guidelines in the American Ornithologists' Union Report of Committee on the Use of Wild Birds in Research (American Ornithologists' Union 1988) and the protocol was approved by the Auburn University Institutional Animal Care and Use Committee, IACUC.

We monitored all birds ≥2 times weekly using the homing method (White and Garrott 1990), and recorded all locations on aerial photographs. Specific causes of predation (raptor, mammal, and snake) were determined when possible by the condition of the transmitter and evidence at the kill site (Dumke and Pils 1973, Curtis et al. 1988). Reporting rates for harvest were virtually 100% because these studies were conducted on private property, hunting and harvest were completely controlled, and records of all harvested birds were put into a large research data base (Terhune et al. 2007). When sufficient evidence did not exist to put the cause of mortality into one of these categories, or when evidence existed implicating more than one mortality agent, the cause was recorded as unknown. Any other known sources of mortality (i.e., accidents) were recorded as "other". Known mortalities therefore were classified as raptor, mammal, snake, harvest, and other.

Seasonal and annual survival estimates were based on a biological year beginning 1 October and ending 30 September. This annual period was divided into 2 seasonal intervals for analysis. The fall-winter interval (1 Oct - 31 Mar, 182 days) began with termination of nesting and formation of coveys. The spring-summer interval (1 Apr - 30 Sep, 183 days) began with covey breakup and initiation of nesting. The Kaplan-Meier staggered entry method was used to produce seasonal and annual survival estimates (Kaplan and Meier 1958, Pollock et al. 1989). An effort was made in every case to insure that birds were randomly sampled across the landscape during trapping and radio-tagging (Pollock et al. 1989, White and Garrott 1990). We used the traditional 7-day conditioning period where birds that died or were censored within 7 days of radio-tagging were excluded from the analysis (Kurzejeski et al. 1987, Pollock et al. 1989). We present causes of mortality by season and site as the probability of loss per known-fate individual (Cox et al. 2004, Terhune et al. 2007). Annual survival estimates were subjected to methods employed by Guthery and Lusk (2004) for determining whether they were reasonable and reliable.

Parts of the data presented here have been published previously. Burger et al. (1998) presented survival and cause specific mortality for one of these sites (site 1) for a 5-year period. They reported

Table 1: Years of study, sample size, and number of annual and seasonal survival estimates for radio-tagged northern bobwhite on 13 study sites of the Albany Quail Project in Georgia and Alabama 1992-2005.

			Number of Survival Estimates		
Site	Years of Study	n	Annual	Fall - Spring	Spring - Fall
1	1992-2005	2383	13	13	14
2	1993-2005	1596	10	11	11
3a	1999-2005	671	6	6	7
3b	2000-2005	562	5	5	5
4	2000	50	0	0	1
5	1998-2000	154	2	2	2
6	1999-2002	507	3	4	3
7a	2003-2004	120	0	0	2
7b	2003-2004	120	0	0	2
8a	2003-2005	219	2	2	3
8b	2003-2005	200	2	2	3
9	2002-2005	269	3	3	3
10	2002-2005	254	3	3	3
TOTAL	1992-2005	7105	49	51	59

higher seasonal and annual survival estimates than other previous studies throughout the Southeast. (Sisson et al. 2000*a,b*) working on another of these sites (site 2) reported higher survival in some seasons for birds receiving supplemental feed, and lower survival for wild birds where a large influx of pen-raised birds occurred. Hughes et al. (2005) compared survival estimates over a 2-year period for 2 of these sites (site1 and 5) which revealed higher survival on the site being managed for quail than on a farm landscape. Terhune et al. (2007) reported 2 summer survival estimates for site 1 and compared these to wild birds relocated on this site. Together these estimates make up 7 of the 49 (14%) annual estimates presented here and 19 of the 110 (17%) seasonal estimates. They are included here for comparative purposes, to provide a more comprehensive report on survival estimates, and to help derive long term averages for each site. Because our study was observational we focus on means, standard errors, confidence intervals, and graphical interpretation of results to compare among sites and years.

Results

From 1992-2005 we monitored 7,105 radio-tagged bobwhites that survived the 7 day censor period. The years of study, number of radio-tagged individuals, and the number of seasonal and annual survival estimates generated for each site is presented in Table 1. Continual radio-telemetry monitoring yielded 49 separate annual (Oct - Sep) survival estimates. Annual survival for all sites and years combined averaged 0.196 (SE = 0.011; range: 0.08-0.40). We observed annual survival rates sufficient to maintain stable populations (defined as production [juveniles/adult] required to stabilize the population ≤ 7) for 38 of these 49 (78%) individually derived annual survival estimates (Figure 2). Average long-term annual survival by site ranged from

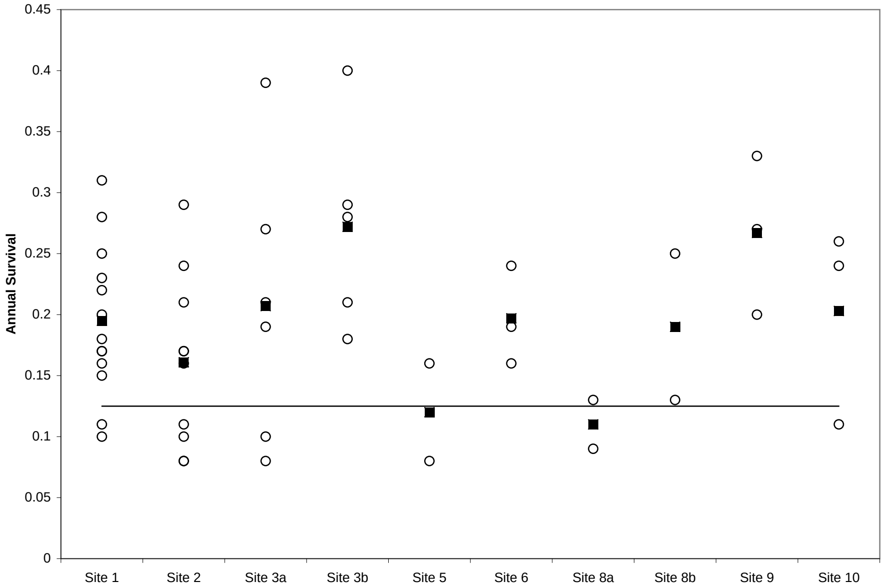

Figure 2: Annual estimates (Kaplan-Meier) of survival for northern bobwhites on 10 study sites of the Albany Quail Project in Georgia and Alabama 1992-2005. The darkened point represents the mean of the annual estimates for each site. The horizontal line represents annual survival of 0.125 below which necessitates a stabilizing age ratio ≥7 for population stability.

0.11 (SE = 0.028) to 0.272 (SE = 0.031; Table 2) with a mean (n = 10) of 0.192 (SE = 0.017). Eight of these 10 (80%) were deemed sufficient to maintain stable populations (Figure 2). Over-winter seasonal survival estimates (n = 51) averaged 0.541 (SE = 0.019; Table 2) and ranged from 0.25 - 0.82 (Figure 3); while breeding season survival estimates (n = 59) averaged 0.352 (SE = 0.013; Table 2) and ranged from 0.13-0.59 (Figure 4).

The cause of mortality was determined from 3,580 known fate individuals and was unknown for an additional 801. Over-winter seasonal mortality (n = 1,473) was attributed to avian predation (0.572 ± 0.040), mammal predation (0.265 ± 0.044), harvest (0.156 ± 0.028) snake predation (0.001 ± 0.004) and other (0.005 ± 0.002; Table 3). Breeding season mortality (n = 2,138) was ascribed to avian predation (0.613 ± 0.026), mammal predation (0.339 ± 0.049), snake predation (0.037 ± 0.006), and other (0.011 ± 0.004; Table 3).

Discussion

We begin by discussing the reliability of our telemetry based survival estimates. The reliability of information collected via radio-telemetry has recently been called into question by Guthery and Lusk (2004) based on their review of the literature and studies in Oklahoma (Cox et al. 2004). This has been countered more recently by researchers in the Southeast however, based on long-term studies and large sample sizes (Terhune et al. 2007, Palmer and Wellendorf 2007). Guthery and Lusk (2004) posited that annual survival estimates from telemetry were biased low because only 10 of 58 (17%) years in the studies they reviewed produced estimates they considered reasonable. The point was reiterated by Cox et al. (2004) where estimates considered reasonable were obtained through telemetry in only 1 of 10 (10%) years for their study in Oklahoma. The benchmark for a reasonable estimate used during these studies was a stabilizing age ratio ≤7, which was considered the upper acceptable limit of production (juvenile/adult) required to stabilize the pop-

Table 2: Over-winter (1 October - 31 March), breeding season (1 April - 30 September), and annual Kaplan-Meier survival estimates on 10 study sites of the Albany Quail Project in Georgia and Alabama 1992-2005.

	Over-winter		Breeding Season		Annual	
Site	S	SE (S)	S	SE (S)	S	SE (S)
1	0.467	0.022	0.411	0.023	0.195	0.017
2	0.483	0.036	0.324	0.022	0.161	0.022
3a	0.618	0.051	0.296	0.055	0.207	0.047
3b	0.686	0.017	0.396	0.059	0.272	0.031
5	0.305	0.055	0.385	0.055	0.120	0.040
6	0.570	0.023	0.327	0.033	0.197	0.023
7a			0.405	0.020		
7b			0.390	0.015		
8a	0.400	0.050	0.303	0.054	0.11	0.028
8b	0.635	0.015	0.283	0.050	0.19	0.06
9	0.777	0.034	0.340	0.038	0.267	0.038
10	0.597	0.078	0.330	0.047	0.203	0.047
TOTAL	0.541	0.019	0.352	0.013	0.196	0.011

ulation. This could have also been interpreted as saying any annual survival estimate generated from telemetry $\leq$0.125 was considered unreasonable for a stable population because it was too low for reproduction to overcome. We use this procedure to critique our estimates.

The average annual survival we observed (0.192) necessitated production of only 4.2 juveniles per adult, well below the threshold of 7 proposed by Guthery and Lusk (2004) to maintain population stability. In addition, of the 49 annual survival estimates generated during this study, 38 (78%) were above the 0.125 level, and would therefore have been classified as "reasonable" in their analysis (Figure 2). A more sensible approach may have been to look at the long-term average-annual survival by site as apposed to using each year as a separate data point. Even populations considered stable can be expected to fluctuate from year to year (Palmer et al. 2002, Palmer and Wellendorf 2007) along with the survival and reproductive parameters that drive them. Therefore, we believe looking at the average annual survival for a site over time and the subsequent stabilizing age ratio required for that average is more judicious. When examined in this fashion it is revealed that 6 of 10 sites had individual years with annual survival estimates below 0.125, however the average for 8 of 10 of these sites (80%) was above the threshold of what is believed needed to maintain population stability (Table 2 and Figure 2). Incidentally, both of our study sites (site 5, 8a) where the average estimates would be classified as "unreasonable" were from properties with declining populations during the course of a short-term investigation. Site 5 was an unmanaged farm where canopy closure on young pine stands shaded out the ground cover and reduced habitat quality and quail abundance (Hughes et al. 2005, Terhune et al. 2007). This site was typical of the "non plantation" landscape in southwest Georgia and had no harvest pressure. An-

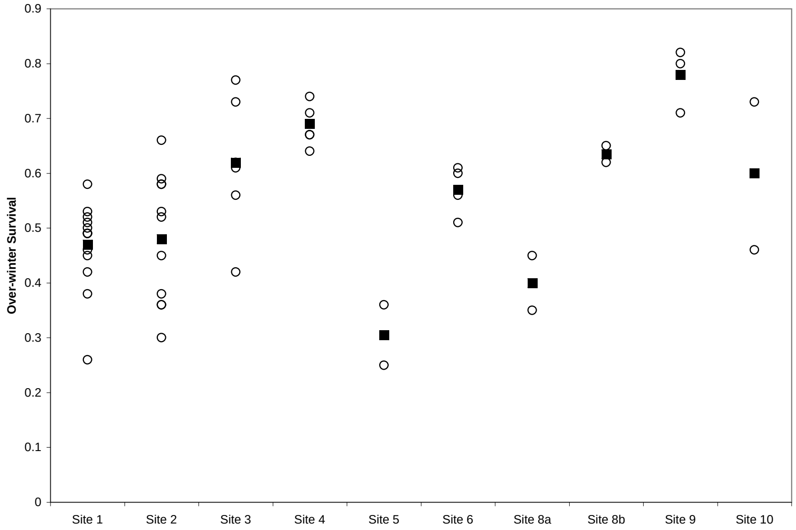

Figure 3: Over-winter (1 Oct - 31 Mar) seasonal estimates (Kaplan-Meier) of survival for northern bobwhites on 10 study sites of the Albany Quail Project in Georgia and Alabama 1992-2005.

nual survival on this site was shown to be a function of over-winter survival which in turn was affected by the abundance of native foods (Hughes et al. 2005). Site 8 was a managed property where unusually high avian mortality from Cooper's Hawks (*Accipiter cooperii*) during winter caused a population decline as evidenced by our census records (AQP, unpublished data) as well as the plantations hunting records. The other 8 study sites where "reasonable" estimates come from are all from stable or even increasing populations (Burger et al. 1998, Sisson et al. 2002*b*,*a*, Terhune et al. 2007). We interpret this disparity in telemetry derived survival estimates as evidence of demographic problems in need of management attention, and not evidence that the estimates are "unreasonable" because they are too low.

Additionally, short-term studies may be conducted during either up or down swings in population trajectory thus giving a false impression of the true long-term demographics. To illustrate, we looked at only the study sites with relatively long term samples (≥5 years) from our data. These 4 sites generated 34 of the 49 (69%) annual survival estimates, including 8 (24%) that are below the 0.125 required to maintain a stable population. Guthery and Lusk (2004) analysis would have classified these 8 as unreasonable and evidence of radio-handicapping. Each of these 4 sites had long-term annual survival well above the threshold (avg. = 0.21, range 0.161-0.272) which leads us to interpret these low estimates as expected fluctuations in population performance. In fact, population abundance estimates have been shown to fluctuate along with these fluctuations in survival (Sisson et al. 2000*b*, Palmer and Wellendorf 2007).

As a result of this evidence and in combination with the results reported by others (Palmer and Wellendorf 2007, Terhune et al. 2007), we conclude these telemetry based survival estimates are indeed reliable for our study sites. We continue our discussion based on that conclusion.

The survival rates reported herein for these sites are generally higher than many of the previously reported rates for the Southeast, and tend to support and strengthen the arguments of Burger et al. (1998) that these managed populations perform better, demographically, than their counterparts on the typical landscape of the Southeast. The disparity is particularly apparent in over-winter survival when these estimates are compared to the review conducted by

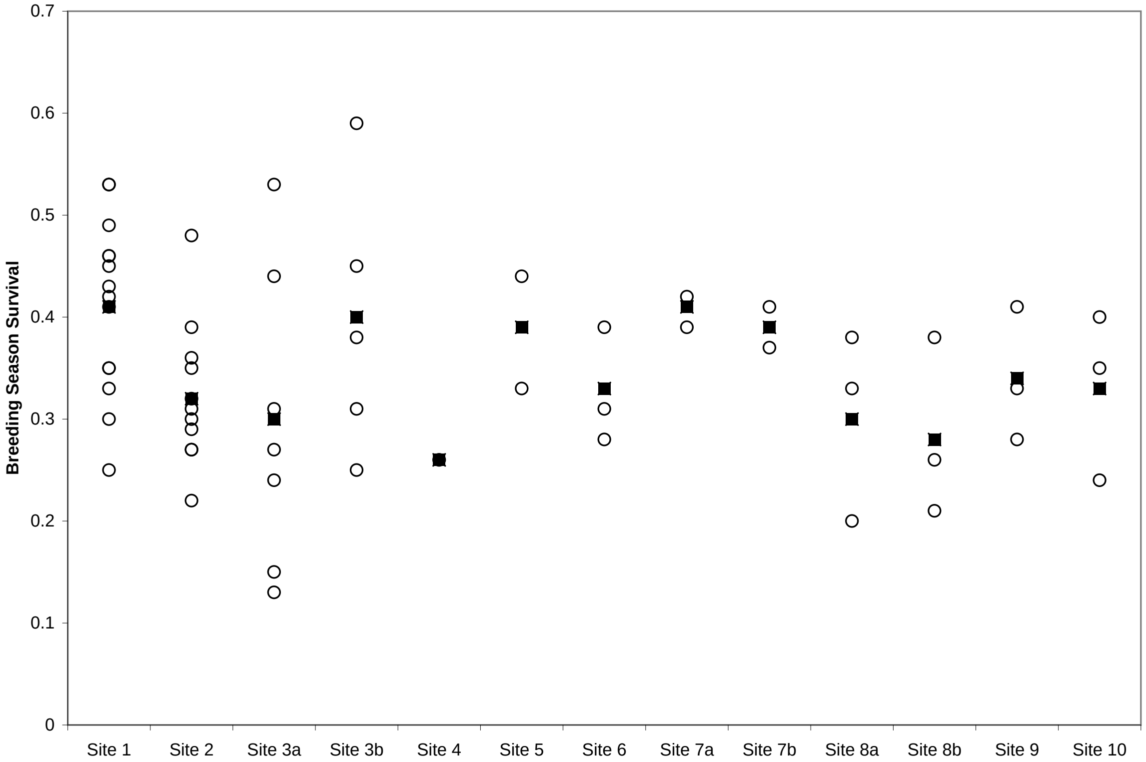

Figure 4: Breeding season (1 Apr - 30 Sep) seasonal estimates (Kaplan-Meier) of survival for northern bobwhites on 13 study sites of the Albany Quail Project in Georgia and Alabama 1992-2005.

Guthery and Lusk (2004) and the 10-year study on annual survival by Cox et al. (2004). During these studies, low annual survival was generally a function of low over-winter survival from a combination of predation and harvest. On our study sites, harvest was generally low (<10% Burger et al. 1998, Terhune et al. 2007) and the birds resided in a landscape of good habitat where supplemental feed was provided in most cases. Most of the other studies conducted in the Southeast were conducted during a time of population decline (Sauer et al. 2004) caused by diminishing habitat and increasing predator populations (Brennan 1991, Rollins and Carroll 2001). Combine this with the possibility of diminished food resources due to clean farming, less prescribed burning, and increased deer browsing on native food plants (Stokes et al. 1994) it is easy to imagine why many of these other studies reported low over-winter survival. Exceptions to this included Curtis et al. (1988) and Pollock et al. (1989); who both reported over-winter and annual survival estimates similar to ours on managed and un-hunted populations in north Florida. These patterns tend to support the conclusion of Cox et al. (2004) of accumulating evidence that harvest may increase over-winter mortality and decrease breeding populations.

In our study, over-winter survival (0.541) was generally higher than breeding season survival (0.352; Table 2). Breeding season survival as reported herein was similar to many of the previously published studies in the region (Curtis et al. 1988, Puckett et al. 1995, Taylor et al. 2000, Hughes et al. 2005, Terhune et al. 2007) suggesting the major difference in regional demographics was a higher percentage of the population in our study making it to the breeding season. Causes of mortality generally follow the trends observed in previous studies. For all our study sites combined avian predation ac-

Table 3: Over-winter (1 October - 31 March) and breeding season (1 April - 30 September) causes of known mortality for 3,611 radio-tagged northern bobwhite on 10 study sites of the Albany Quail Project in Georgia and Alabama 1992-2005.

	Causes of Mortality					
Season	Avian	Mammal	Harvest	Snake	Other	Total
Over-winter						
Total	843	391	229	2	8	1473
Percent	57.23	26.54	15.55	0.14	0.54	100
Breeding						
Total	1310	725	0	80	23	2138
Percent	61.27	33.91	0	3.74	1.08	100

counted for the highest percentage of known mortalities in both winter (0.572) and summer (0.613; Table 3). Mammal predation was the next most important mortality factor and was higher during the spring-summer (0.339) than fall-winter (0.265). The percentage of known mortality attributed to harvest, snakes, and other mortality factors was inconsequential on our study sites compared to these two agents (Table 3). This is in stark contrast to Curtis et al. (1988), Robinette and Doerr (1993) and Cox et al. (2004) who each reported high winter mortality from a combination of avian predation and heavy harvest pressure. Harvest pressure was light (<10%) on all of our study areas and likely helped promote the high over-winter survival estimates.

While our survival estimates were generally higher than most previously reported studies in the Southeast, there was still some variability in both the annual and seasonal estimates (Figure 2 - 4). This was true when compared between sites and between years on the same site. We believe this is partly due to the natural fluctuation of quail populations as discussed previously as well as to site-specific management problems between properties. Several sites with long-term-stable populations have separate years, or series of consecutive years, with low annual survival estimates; while the 2 sites with population problems (5,8a) have consistently low annual survival due to low over-winter survival. These parameters have proven to be a valuable tool for us in developing site-specific management plans as well as in contributing to the knowledge base of quail demographics for the region to aid in species recovery. We further suggest caution when interpreting survival rates of short-term studies as they may provide false impressions of what the population may permit during the long-term.

Management Implications

One of our intentions was to dispel the notion that all telemetry based survival information was biased low and therefore unreliable. We disagree that research biologists should be skeptical of information obtained on bobwhite demographics obtained with radio telemetry, but only after the researchers themselves have demonstrated the reliability of the data for their study sites. We caution, however, in drawing too many conclusions from short-term and/or small sample size studies. We have observed that when our estimates are derived over long term and from large samples they are reliable, indicative of what the population is doing, and very helpful in making site-specific management recommendations. Perhaps on a regional scale, we should focus more on what previously published low survival estimates are telling us instead of using them as evidence of "radio-handicapping".

The fact that our survival estimates are higher than the general landscape of the Southeast is no surprise because our study was conducted mostly on properties managed intensively for bobwhite quail with a conservative harvest. However, this inherent management and moderate harvest rate does not preclude the utility of our research from benefiting other sites. Rather, we believe that many lessons can be gleaned from the management programs that yield this high survival (i.e. timber thinning, prescribed burning, hardwood removal, supplemental feeding, and predation management) that may contribute to region-wide species recovery efforts. In short, sound management practices at the landscape-scale level results in improved demographic parameters and population persistence.

Acknowledgments

We are extremely grateful to the landowners involved in this study for allowing us to use their properties as outdoor laboratories as well as for their generous financial support of our work. The plantation managers and their staff have been invaluable to us over the years and we appreciate them. We also thank the many financial contributors to the Albany Quail Project; we could not do this without them. Dr. Dan Speake was one of the founders of this work and we appreciate him for his contribution as well as those of all the graduate students and staff of the AQP over the last 14 years. This project has been a real team effort and it has been a distinct pleasure to have lived and worked in such a unique part of the world.

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Techniques

Necklace-type Transmitter Attachment Method for Ruffed Grouse Chicks

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Although methodologies to obtain cause-specific mortality and survival information for adult ruffed grouse (*Bonasa umbellus*) are well documented, procedures for determining similar parameters are lacking for grouse chicks. Mortality among grouse chicks is believed highest during the first few weeks posthatch. During 1999-2002, we equipped ruffed grouse chicks ($n = 97$) from 33 separate broods, $\leq$4-days-old with radio transmitters to assess the efficacy of transmitters and to examine survival/mortality. Further, we observed that grouse chicks retained transmitters (100%) until recapture or mortality. Handling time was limited because transmitter attachment took only a few minutes per brood. We observed mortality fates for 91% of radio-collared chicks. Therefore, because of the non-intrusive nature, field application, and retention of necklace-style transmitters employed in this study, this method may provide a desirable alternative to assessing survival/mortality among ruffed grouse chicks.

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Key words: *Bonasa umbellus*, necklace-type collar transmitter, ruffed grouse chicks, West Virginia

Introduction

Mortality in ruffed grouse (*Bonasa umbellus*) is thought to be highest during the first few weeks of life (Rusch et al. 1984, Bergerud 1988), but factors influencing chick survival are not well documented. Survival and cause-specific mortality are important components in population management. Grouse chick mortality/survival estimates are often derived from flush counts. Estimating chick numbers via flush counts in <2-week-old broods, however, is highly unreliable (Larson et al. 2001). Although survival estimates and mortality causes of adult ruffed grouse can be readily obtained via radio telemetry (Godfrey 1975, Maxson 1977, 1978, Small et al. 1991), transmitter size and attachment methods have limited examination of these parameters for young ruffed grouse chicks.

Several methods of transmitter attachment have been used to study young gallinaceous birds including glue-on (Bowman et al. 2002, Spears et al. 2002), subcutaneous anchor (Mauser and Jarvis 1991), and elastic harnesses (Peoples et al. 1995, Hubbard et al. 1998) on domestic and wild turkey (*Meleagris gallopavo*) poults, and interscapular implants on ring-necked pheasant (*Phasianus colchicus*) chicks (Ewing et al. 1994, Riley et al. 1998) and turkey poults (Korschgen et al. 1996, Hubbard et al. 1999, Bowman et al. 2002). Only a few telemetry studies of ruffed grouse chicks have been conducted. Larson (1998) and Larson et al. (2001) attached transmitters via interscapular implants or external suture technique to 6-day-old (mean 6.4 days, range 5-10 days) ruffed grouse chicks in Michigan. Similarly, Burkepile et al. (2002) used a suture technique to attach transmitters to 1-day-old sage grouse (*Centrocercus urophasianus*) in Idaho. Because of the diminutive size and mass of ruffed grouse chicks relative to sage grouse, the potential effects of transmitter mass (i.e., >5% transmitter to body mass ratio) and the intrusive nature and stress of the procedure on the animal are important considerations.

Subcutaneous interscapular implants are intrusive and require a sterile environment to reduce in-

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fection rate (Korschgen et al. 1996). Hubbard et al. (1998) found them to affect growth in wild turkey, although the long-term effects are unknown (Hubbard et al. 1998); however, no effect on wing growth was reported in domestic turkeys (Bowman et al. 2002). Wild turkey poults with interscapular implants and mallard (*Anas platyrhynchos*) ducklings with subcutaneous anchors exhibited short-term (2-4 hour) loss of balance and difficulty in walking (Mauser and Jarvis 1991, Bowman et al. 2002). Glue-on attachment of transmitters has been used successfully for turkey poults with no apparent effect on growth or survival, although retention time may be limiting (<29 d) for some applications (Bowman et al. 2002, Spears et al. 2002). Necrosis at the site of attachment has been found when using cyanoacrylate glue to retain transmitters (Burkepile et al. 2002).

As part of a ruffed grouse population ecology study, we wanted to examine survival and mortality of chicks from hatch to age 5 weeks. We were particularly interested in the period from 0-2 weeks posthatch. Because it has been suggested that ruffed grouse chicks exhibit high mortality rates during the first few weeks posthatch (Rusch et al. 1984), it is important to monitor chicks as early as possible while also minimizing capture- and transmitter-related stress (Caccamise and Hedin 1985, Dobony 2000). To minimize our influence on mortality, we selected an attachment method that was non-intrusive, could be attached in the field, and did not require extended periods of recovery or involve excessive handling time. Although successful in other studies (Bowman et al. 2002, Spears et al. 2002), our preliminary experience with glue-on transmitters was unacceptable. We chose to not use the external suture method described by Larson et al. (2001) because of its intrusive nature and our desire to monitor chicks <6 days old. We report on the development of a necklace-type transmitter for use on young ruffed grouse chicks. Our objective was to examine the efficacy of this method on ≤4-day-old chicks and its usefulness in assessing mortality through age 5 weeks.

Study Area

From 1999-2002, we conducted research on ruffed grouse on the MeadWestvaco Ecosystem Research Forest (MWERF) located in Randolph County, West Virginia, and situated in the Allegheny Mountain physiographic province (Fenneman 1938). In 1999, we also used grouse chicks from the MeadWestvaco Dutch Run Tract (DRT) located in Greenbrier County, West Virginia, classified as part of the Ridge and Valley physiographic province (Fenneman 1938).

The 3,413 ha MWERF was established by Westvaco Corporation in 1994 to study industrial forestry impacts on ecosystems and their processes. MeadWestvaco managed the MWERF for forest products, and its oldest forests were second-growth stands established after harvests at the turn of the 20th century (Tilghman 1989, Clarkston 1993). MeadWestvaco managed stands on 40-80 year rotations depending on site characteristics and quality. Harvest methods included diameter-limit to remove valuable sawtimber as well as clearcut and deferment harvests (i.e., shelterwood harvest or clearcut with reserves) for stand regeneration. Elevations ranged from 740-1200 m and topography was rugged, with plateau-like ridgetops atop steep slopes and narrow valleys (Fenneman 1938, Ford and Rodrigue 2001). The MWERF was characterized by a cool, moist climate with average annual precipitation exceeding 198 cm (http://www.nndc.noaa.gov). Forest cover primarily was Allegheny hardwood-northern hardwood, mixed mesophytic or cove hardwood associations typical to the Allegheny Mountain physiographic province (Eyre 1980).

MeadWestvaco managed the 2,036 ha DRT strictly for fiber production by clearcutting on an even-aged rotation length of 40-70 years. DRT had a lower site index and received less annual precipitation (107cm; http://www.nndc.noaa.gov) than the MWERF. Elevations ranged from 520-1100 m and topography was extremely steep and rugged with ephemeral seeps and streams throughout. Forest cover was primarily oak-hickory associations

(94%) typical of the Ridge and Valley physiographic province (Eyre 1980).

Methods

Trapping and Monitoring Females

We trapped adult ruffed grouse in fall 1998 and spring 1999 using modified lily-pad traps (Gullion 1965). Once captured, we weighed, aged and sexed (Kalla and Dimmick 1995), and tagged each bird with an aluminum leg band (#12 butt-end tags, National Band and Tag, Newport, KY). We also equipped each female with a necklace-type radio transmitter (Advanced Telemetry Systems, Isanti, MN; multiple models were used through the course of the study). Transmitters weighed 10-11 g, had a 2-year battery life, and were equipped with a motion-sensitive mortality sensor.

After release, we monitored radio-marked females twice weekly using a 2-element Yagi antennae and portable receiver (Wildlife Materials, Carbondale, IL, and Advanced Telemetry Systems, Isanti, MN). Beginning 1 March, we monitored females 3 times weekly to accurately determine nest initiation. We obtained azimuths from permanently located global positioning system telemetry stations and grouse locations [mean bearing error and linear error: 7 and 76 m, respectively (Whitaker 2003)] were determined through triangulation (Mech 1983). As the nesting and breeding seasons progressed, we used triangulation and homing techniques to find nest sites (Mech 1983). After locating nests, we obtained at least 2 egg counts by either flushing females from their nests or counting eggs while nests were unattended; one count occurred during egg laying (if found in time) and one during incubation. We used this information to predict hatch dates (i.e., by backdating to when the last egg was laid) and to determine maximum number of potential chicks per brood.

Capturing and Radio-marking Chicks

We randomly selected broods to equip with radio transmitters. We located broods within 24-48 hours posthatch by homing the female's telemetry signal (Mech 1983). We approached females' locations (<20 m) as quickly as possible to discourage them from hiding chicks and leading us away from broods. We tried not to flush females until we were close enough to easily locate and capture the brood. Once females had flushed, we immediately stopped to avoid trampling unseen chicks and each person captured any chicks that came towards them. We assumed that the potential number of chicks available for capture to be the number of hatched eggs present in the nest. We often located unseen chicks by contact calls they made to females. We captured as many brood members as possible, as quickly as possible, and then carefully moved a short distance away to process the brood (i.e., weigh, affix transmitter) into an area where we were confident no chicks were present. We placed those chicks we captured into a soft fabric bag until fully processed. Handling time for each brood did not exceed 15 minutes post-capture. All chicks (radioed and non-radioed) were released at the capture site after which we immediately vacated the area to allow females to gather broods. All broods were captured between 1000 and 1400 hours and we postponed brood captures if poor weather conditions threatened.

In 1999, we captured chicks 2-3 days posthatch, weighed each individual, and randomly selected chicks from each brood to receive transmitters. Necklace-type transmitters (model MD-2CT; Holohil Systems Ltd., Ontario, Canada) weighed 0.98 g (approximately 7-8% of the body weight at time of attachment) and had a 5-week battery life (Figure 1). We used necklaces made of polyethylene tubing used in arterial surgery (Intramedic Clay Adams Brand , Sparks, MD) that initially had 26 mm circumference loops, but we later increased loop-size to 32 mm (Dobony 2000). We placed monofilament fishing line (2.7-kg test) inside the tubing and knotted it to secure the necklace; we further secured knots with glue formulated especially for monofilament (Anglin' Glue , Clemence Inc., Alpharetta, GA). Numbers of radio-marked chicks per brood ranged from 1-5 depending on numbers of chicks we captured per brood and overall brood size. We

Figure 1: Ruffed grouse chick with necklace-type transmitter attachment method implemented on the MeadWestvaco Ecosystem Research Forest in Randolph County, West Virginia and the MeadWestvaco Dutch Run Tract in Greenbrier County, West Virginia in 1999-2002.

censored chicks that died during the first 24 hours post-release.

In 2000-2002, we slightly modified our transmitter procedures. Upon capture at 2-4 days posthatch, we attached modified necklace-type transmitters (model BD-2A) that weighed 0.54 g, had a 3-week battery life, and had necklaces 42 mm in circumference. The transmitters were configured in the same style as model MD-2CT and attached similarly. Changing transmitter models allowed us to stay within the 5% transmitter to body mass ratio during the first week.

Monitoring Females and Broods

We monitored female grouse and their broods ≥ 1 time per day, typically <2 hours after sunrise and <2 hours before sunset each day. We determined brood locations via triangulation of females' telemetry signals. We then approached the female (usually to within 150 m) and took azimuths on each chick. For chicks not in close proximity to the female, we attempted to retrieve lost chick(s), transmitter(s), or both. We examined all remains for cause of death and performed necropsies if the immediate cause of death could not be determined.

In 2000 and 2001 we conducted 3- and 5-week brood flush counts of all radio-collared females, which included both broods with transmittered chicks and broods in which no chicks were marked. We estimated brood sizes by locating females via telemetry and flushing broods to make an ocular estimate of chick numbers. We considered this a minimum estimate of brood size. We performed no statistical comparison of means, however, because collared and uncollared broods received different levels of disturbance. We captured collared broods at least twice (i.e., to affix initially, replace at 2 weeks posthatch, or remove collars) depending on survival, whereas uncollared broods were never captured. Therefore, we only provide mean ($\pm SE$) number of chicks per brood at 3- and 5-week flush counts and not statistical comparison of means.

Recapturing Chicks

In 1999, we recaptured radio-collared chicks at 2 weeks posthatch and replaced each collar with one that had a necklace circumference of 52 mm or increased the circumference on the existing collar to allow for growth. We recaptured chicks at 5 weeks posthatch to remove collars. Because chicks retained necklace-type transmitters, we were able to find them via their telemetry signal. If flushed, chicks usually flew only short distances and hid. Once hidden, grouse chicks tend not to move and we easily captured them by hand. When capturing older chicks, we again took care to avoid trampling unmarked chicks. After replacing collars, we returned chicks to where we had flushed them initially and immediately left the area.

In 2000-2002, we again recaptured radio-collared chicks at 2 weeks posthatch and replaced 3-week transmitters with model MD-2CT transmitters (0.98 g, 5-week battery life) that had 52 mm necklaces. This allowed us to reliably track chicks for the 5-week period and accommodate for rapid growth in ruffed grouse chicks. All handling procedures were approved and conducted under West Virginia University's Animal Care and Use Committee protocol 01-0405.

Results

In 1999, we captured 55 chicks from 10 broods (6 at MWERF, 4 at DRT) within 72 hours posthatch. We equipped 35 of the 55 chicks (20 at MWERF, 15 at DRT) with modified necklace-type transmitters. Chicks weighed 12.9 ± 0.2 g (mean ± SE, $n = 35$; range = 11.4-15.7 g) upon initial capture. From 2000-2002, we captured 86 chicks from 23 broods within 96 hours posthatch on the MWERF. We equipped 62 of 86 chicks with necklace-type transmitters. Chicks selected to receive radio transmitters weighed 14.8 ± 0.3 g (mean ± SE, $n = 62$; range = 11.2-21.2 g) upon initial capture. We released all chicks within 15 minutes of each capture.

All chicks marked during 1999-2002 retained their transmitters throughout the 5-week posthatch sampling period or until death. We were able to determine fates of 88 of 97 (91%) of radio-collared chicks. All chicks ($n = 22$) surviving 9-14 days were successfully recaptured to adjust necklace circumferences or replace transmitters. Of these, five reached 35 days posthatch and were recaptured to remove their transmitters. Transmitter-related mortality decreased from 38% (11 of 29 known mortalities) during our initial field season in 1999 to 8% (3 of 40 known mortalities) during 2000-2002. Ocular brood size estimates between collared and uncollared broods were similar within 3- and 5-week flush counts in 2000 and 2001 (Table 1). However, we performed no statistical comparison of means because collared and uncollared broods received different levels of disturbance (e.g., uncollared broods were never captured whereas collared broods were captured twice).

Discussion

Our objective was to develop a transmitter attachment method for ruffed grouse chicks that (1) was non-intrusive, (2) could be completed in the field, (3) did not require an extended period of recovery or involve excessive handling time, and (4) would be retained by the animal throughout the focal period (i.e., 0-5 weeks). Our goal in meeting these criteria was an attachment method that minimized anthropogenic influence while enabling assessment of cause-specific mortality.

Dobony (2000) first used our method in 1999 but found that the necklace circumference required refinement. Rapid growth of ruffed grouse chicks made it difficult to predict what circumference to initially make the necklace in 1999, as well as what circumference was needed at 2 weeks posthatch. The necklace had to be snug enough to prevent chicks from getting their beaks or feet caught, but also had to allow for passage of food items. Dobony (2000) reported that several chicks died in 1999 after ingesting terrestrial snails, which were too large and rigid to pass below the necklace. This resulted in the 38% transmitter-related mortality rate we report for 1999. Enlargement of the necklace circumference used for 0-2-week-old chicks from 34 to 42 mm remedied this

Table 1: Mean (± SE) number of ruffed grouse chicks observed at 3- and 5-week flush counts of radio-collared broods and broods that did not receive radio collars on the MeadWestvaco Research Forest, West Virginia, 2000-2001. Number of broods is in parentheses.

	Year	
	2000	2001
3-week estimates		
Collared broods	(6) 1.67 ± 0.56	(8) 3.10 ± 0.55
Uncollared broods	(3) 1.67 ± 1.20	(4) 2.00 ± 1.08
5-week estimates		
Collared broods	(5) 1.60 ± 0.67	(6) 2.00 ± 0.55
Uncollared broods	(3) 1.00 ± 0.58	(9) 1.30 ± 0.47

problem and contributed to the substantial decrease in transmitter-related deaths recorded during 2000-2002.

Transmitters we placed on ≥3-day-old chicks were approximately 7-8% of their body mass in 1999, while in 2000-2002 the transmitters were <5%. Typically the transmitter to body mass ratio rule-of-thumb has been <5%. This is often associated with birds that have the stress of flight (Caccamise and Hedin 1985, Brigham 1989). However, ruffed grouse chicks do not fly until 4-5 days old, and then may fly only short distances. They are physically unable to fly longer distances until their flight feathers have developed. By this time, rapid growth has quickly decreased the transmitter to body mass ratio (Speake et al. 1985). Mauser and Jarvis (1991), Mauser et al. (1994), and Davis et al. (1999) found no effect on survival in ducklings when using transmitters weighing 5-7% of body mass. Speake et al. (1985) placed transmitters on turkey poults that weighed approximately 6% of the body mass and found no impact on survival. To further alleviate any concerns of transmitter mass on survival, we used smaller 0.54 g radio transmitters for the first 2 weeks posthatch in 2000-2002 and replaced them with 0.98 g transmitters for the remaining 3 weeks.

Our brood flush count estimates for 2000-2001 provide limited support that transmitters had minimum influence on chick survival through 5-weeks posthatch. We found 3- and 5-week flush counts appeared similar between collared and uncollared broods. Collared broods at 3- and 5-week flush counts had more chicks per brood on average than did uncollared broods, despite a greater level of disturbance (note: simple comparison only; no statistical comparison of means performed). Such low brood counts - although indicative of higher rates of mortality than commonly reported in other parts of the range - are similar to those found in the central Appalachian region (Haulton 1999, Devers 2006, Smith 2006). Flush counts have been criticized for producing biased estimates of chick numbers (Godfrey 1975). However, presence of such bias in our estimates would not negate support for minimum transmitter influence because our 3- and 5-week flush count protocol was consistent between collared and noncollared broods.

Our 100% transmitter retention until recapture or death provided mortality fates for 91% of radio-collared chicks. Because the transmitter was firmly attached around the chick's neck, predators had to either expend more effort to remove the transmitter (thus leaving teeth marks, beak impressions, bent antennas), or consume the transmitter with the chick (thus allowing the transmitter to show up in scat or pellets). In 3 instances, we located transmittered chicks that were taken to nest sites and fed to nestling hawks. Only chicks whose transmitter apparently failed or was destroyed could not be assigned a fate.

A possible concern with our necklace-type transmitter is the necessity to recapture chicks at approximately 2-weeks posthatch to replace transmitters, and then again at 5-weeks to remove transmitters. Failure to remove the transmitter would result in the death of the chick. We were successful in capturing all candidate chicks at 2- (9-14 days) and 5-weeks posthatch. When flushed, chicks usually flew only short distances (even up to 5 weeks of age) and hid. Once hidden, chicks tended not to move and were easily captured by hand without harm and replacement of the smaller transmitters typically took 10-15 minutes per brood.

Although radio telemetry is the most reliable method for determining timing and extent of mortality and survival (Korschgen et al. 1996), it is important to ensure that transmitters and attachment methods have minimal effect. Our method allowed us to attach transmitters in the field, minimized our handling time, and did not involve subcutaneous implantation, removal of feathers to apply adhesive, or suturing of any kind. Moreover, our necklace-type transmitter allowed us to begin monitoring 2-4 days earlier in the first week posthatch than methods previously described. Because of its use on younger chicks, non-intrusive nature, field application, and retention time, our necklace-type transmitter may provide a desirable alternative to assessing mortality/survival among ruffed grouse chicks.

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A Method for Determining Asymptotes of Home-Range Area Curves

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Home-range area curves are used to estimate the number of locations needed to accurately estimate home range size based on the asymptote of the curve. However, the current methodology used to identify asymptotes for home-range area curves is largely subjective and varies between studies. Our objective was to evaluate the use of exponential, Gompertz, logistic, and reciprocal function models as a means for identifying asymptotes of home-range area curves. We radio monitored northern bobwhite (*Colinus virginianus*) coveys during mid-September through November 2001-2002 in Jim Hogg County, Texas. We calculated home-range size of radiomarked coveys using the 95% fixed kernel with least squares cross validation and minimum convex polygon estimators. We fitted area observations and coefficient of variation to the number of locations using exponential, Gompertz, logistic, and reciprocal function models to estimate the minimum number of locations necessary to obtain a representative home range size for each home range estimator. The various function models consistently provided a relatively good fit for home range area curves and coefficient of variation curves ($0.58 \leq R^2 \leq 0.99$; $P < 0.05$) for both home range estimators. We used an information-theoretic framework (AICC) to select the best model to estimate area-curve asymptotes. The use of function models appears to provide a structured and useful approach for calculating area-curve asymptotes. We propose that researchers consider the use of such models when determining asymptotes for home-range area curves and that more research be conducted to validate the strength of this method.

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Key words: area curves, home range, kernel estimator, minimum convex polygon, northern bobwhite

Introduction

Home-range size, hereafter home-range, is a parameter commonly reported in many radiotelemetry studies (Garton et al. 2001). Home-range is affected by factors such as time elapsed between consecutive locations (Swihart and Slade 1985*a*,*b*), techniques used to collect location data (Adams and Davis 1967), and the number of observations used to obtain the estimate (Stickel 1954, Jennrich and Turner 1969, Bekoff and Mech 1984, Seaman et al. 1999). Several studies have attempted to provide guidelines for calculating home-range by comparing the performance of home range estimators under varying sample sizes (Boulanger and White 1990, Worton 1995, Seaman and Powell 1996, Seaman et al. 1999). However, results have been disparate (Kernohan et al. 2001).

Home-range area curves have been used to estimate the number of locations necessary for estimating home range (Odum and Kuenzler 1955, Bond et al. 2001, Gosselink et al. 2003). A home-range area curve for a species plots the number of independent locations on the x-axis against the estimated home-range size on the y-axis for that particular sample size. From the resulting graph, the number of required locations is denoted when increasing the number of locations does not result in an increasing home range size (i.e., the asymptotes of the graph; Odum and Kuenzler 1955). However, the methodology used to identify asymptotes for home-range area curves is largely subjective and varies between studies. For example, Odum and Kuenzler (1955) defined an asymptote as being the point when additional locations produced <1% change in mean

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home range size, whereas Bond et al. (2001) identified asymptotes through visual inspection. Given this subjective and discordant approach, a more structured methodology is needed to determine the optimum number of locations necessary to produce a representative home range.

The objective of our study was to evaluate the use of exponential, Gompertz, logistic, and reciprocal function models as a means for identifying asymptotes of home-range area curves (i.e., area-curve asymptotes). We used radio locations obtained from radio marked northern bobwhites (*Colinus virginianus*; hereafter bobwhites) to develop home-range area curves and evaluate our proposed methodology.

Study Area

We conducted our radiotelemetry study on a private ranch located 8 km east of Hebbronville, Texas in Jim Hogg County. The study area is contained within the Rio Grande Plains ecoregion (Gould 1975). Topography within the Rio Grande Plains is level to rolling, and the elevation ranges from sea level to 330 m. The Rio Grande Plains is characterized by rangeland, open prairies with a growth of mesquite (*Prosopis glandulosa*), huisache (*Acacia smallii*), granjeno (*Acacia berlandieri*), and Texas pricklypear cactus (*Optuntia lindheimeri*). Annual rainfall ranges from 35 to 66 cm and soils range from clays to sandy loams (Correll and Johnston 1979). Although large acreages of cultivated land exist within the Rio Grande Plains, the predominant land use is livestock production (i.e., rangeland) (Correll and Johnston 1979).

Methods

We trapped bobwhites from mid-August through September 2001 and 2002 using funnel traps baited with milo (Stoddard 1931) and by night netting roosting coveys (Labisky 1968) on 3 pastures (601 ha, 1031 ha, and 1563 ha), each separated by >3 km. We banded all captured bobwhites and radiocollared any bobwhite weighing ≥150 g. We fitted bobwhites with 6-7 g neck-loop radiotransmitters (American Wildlife Enterprises®, Tallahassee, Florida).

We monitored coveys via radiotelemetry 5 times per week from mid-September through November 2001-2002. We defined this 10-week period as the fall season. We located coveys by homing (White and Garrott 1990) and obtained a global positioning system (GPS) coordinate using a hand-held unit with an accuracy of ±5 m (Garmin 90 GPS). We monitored coveys once or twice a day during 1 of 3 time periods: morning (0700-1000 hrs.), afternoon (1200-1500 hrs.), or evening (1600-1900 hrs.). These time periods corresponded to periods of biological activity for bobwhites in southern Texas (i.e., morning feeding, afternoon loafing, and evening feeding, respectively). If 2 locations were taken during the same day for 1 covey, then one location was taken during a loafing period and the other during a feeding period to obtain independent locations. However, if 2 locations were taken during the same day for a specific covey the next location taken for that covey was not taken until 2 days later. For example, if locations were taken on the loafing and evening-feeding period for 1 covey on Monday, then the next location was not taken for the same covey until Wednesday. We followed this procedure in order that covey location is not recorded on the same feeding or loafing site due to temporal autocorrelation of location data.

We calculated home range size of radiomarked coveys using the 95% fixed kernel (Worton 1989) with the least squares cross validation (LSCV) smoothing parameter, and minimum convex polygon (Mohr 1947) home range estimators within the animal movement extension (Hooge and Eichenlaub 1997) of the program ArcView 3.2 (Environmental Systems Research Institute, Inc., Redlands, CA.). We chose to use the kernel home range estimator recommended by Kernohan et al. (2001) because it has the ability to compute home range boundaries that included multiple centers of activity, lacks sensitivity to outliers, is based on complete utilization distribution, and is a nonparametric methodology. We selected the fixed kernel with LSCV because it has lower bias and better surface fit than adaptive kernel

Table 1: Mean home range size (ha), standard error, and coefficient of variation of northern bobwhite coveys over 7 location intervals using the 95% fixed kernel estimator with least squares cross validation (LSCV) smoothing parameter, and minimum convex polygon home range estimator, Jim Hogg County, Texas, USA, Sep-Nov, 2001-2003.

				95% Fixed Kernel (LSCV)			Minimum Convex Polygon		
Year	Location Interval	n[a]	N[b]	Mean	S.E.	CV[c]	Mean	S.E.	CV
2001	Monthly	3	14	20.69	4.55	1.22	1.05	1.02	0.94
	Biweekly	6	14	16.08	4.01	1.01	4.51	2.12	1.17
	Weekly	11	14	17.23	4.15	0.54	8.73	2.95	0.64
	2× Week	20	14	15.32	3.91	0.43	11.06	3.33	0.67
	3× Week	30	14	15.96	3.99	0.36	14.04	3.75	0.5
	4× Week	40	14	15.1	3.89	0.34	14.6	3.82	0.49
	5× Week	50	14	15.02	3.88	0.36	15.6	3.95	0.46
2002	Monthly	3	20	22.84	4.78	0.76	1.42	1.19	1.03
	Biweekly	6	20	11.18	3.34	0.74	2.74	1.66	0.53
	Weekly	11	20	11.34	3.37	0.52	4.69	2.17	0.4
	2× Week	20	20	10.27	3.20	0.42	6.04	2.46	0.29
	3× Week	30	20	11.06	3.33	0.41	8.23	2.87	0.34
	4× Week	40	20	11.34	3.37	0.39	9.45	3.07	0.31
	5× Week	50	20	12.06	3.47	0.41	10.93	3.31	0.3

[a] Number of locations

[b] Number of bobwhite coveys observed

[c] Coefficient of variation

with LSCV for a selected bandwidth (Seaman and Powell 1996, Seaman et al. 1999). We also chose minimum convex polygon because we wanted to assess this commonly used estimator (Seaman et al. 1999).

We developed home-range area curves following a protocol similar to Odum and Kuenzler (1955). We consistently obtained 5 covey locations a week. Based on this schedule we developed separate location intervals to find the minimal number of locations needed to estimate bobwhite home-range size during the fall season. Intervals consisted of 1 location/month, 1 location every other week, 1 location/week, 2 locations/week, 3 locations/week, 4 locations/week, and 5 locations/week, respectively. We calculated mean, standard error, and coefficient of variation (CV) for all covey home range estimates for each location interval. From this data, we then developed home-range area curves (i.e., hereafter area curves) and CV curves for each estimator by year.

Odum and Kuenzler (1955) defined the asymptote as the first location interval at which any additional locations produced <1% change in mean home range size indicating a point of diminishing return. In an attempt to provide a more objective identification of the asymptote, we fitted mean home range size and CV to the number of locations using a exponential, Gompertz, logistic, and reciprocal function models and used an information-theoretic framework (AICC) score to select the best model (lowest AICC; Burnham and Anderson 1998). We used the SAS procedure NLMIXED to run all mod-

Table 2: Model parameters resulting from fitting means of home range size (ha) of northern bobwhite coveys to the number of locations using 4 separate function models. Home ranges were calculated using the 95% fixed kernel least squares cross validation (LSCV) smoothing parameter and minimum convex polygon home range estimators, Jim Hogg County, Texas, USA, Sep-Nov, 2001-2003.

Year *Estimator*		Model parameters			Asymptote						
Model	Function	a	b	C	Estimate (ha)	SE	-1SE	$+1$SE	AIC_C	ΔAIC_C	R^2
2001 *Fixed-Kernel*											
Reciprocal	$f(x) = a + \frac{b}{x}$	14.8	15.90	NA	14.8	0.38	14.5	15.2	29.1	0.0	0.85
Exponential	$f(x) = C + a * e^{(-b*x)}$	-53.0	0.80	15.7	15.7	0.32	15.4	16.0	42.7	13.6	0.86
Logistic	$f(x) = C/(1 + a * e^{(b*x)})$	7.7	0.95	15.7	15.7	0.32	15.4	16.0	42.7	13.6	0.86
Gompertz	$f(x) = 2C - C * e^{(-e^{(a-b*x)})}$	109.3	36.76	15.7	15.7	0.29	15.4	16.0	43.0	13.9	0.85
MCP											
Exponential	$f(x) = C + a * e^{(-b*x)}$	18.1	0.08	15.6	15.6	0.46	15.1	16.1	37.0	0.0	0.99
Reciprocal	$f(x) = a + \frac{b}{x}$	14.6	-45.67	NA	14.6	1.04	13.6	15.6	41.0	4.0	0.89
Logistic	$f(x) = C/(1 + a * e^{(b*x)})$	7.6	0.18	14.8	14.8	0.77	14.0	15.6	48.3	11.3	0.96
Gompertz	$f(x) = C * e^{(-e^{(a-b*x)})}$	8.3	0.82	13.8	13.8	1.08	12.7	14.9	58.7	21.7	0.82
2002 *Fixed-Kernel*											
Exponential	$f(x) = C + a * e^{(-b*x)}$	7.9	2.94	11.2	11.2	0.12	11.1	11.3	37.8	0.0	0.99
Logistic	$f(x) = C/(1 + a * e^{(b*x)})$	4.6	7.49	11.3	11.3	0.16	11.1	11.4	38.2	0.4	0.99
Gompertz	$f(x) = 2C - C * e^{(-e^{(a-b*x)})}$	27.0	6.51	11.3	11.3	0.16	11.1	11.5	38.2	0.4	0.99
Reciprocal	$f(x) = a + \frac{b}{x}$	9.4	34.12	NA	9.4	1.04	8.3	10.4	43.3	5.5	0.77
MCP											
Exponential	$f(x) = C + a * e^{(-b*x)}$	13.5	0.03	14.0	14.0	1.46	12.6	15.5	30.0	0.0	0.99
Gompertz	$f(x) = C * e^{(-e^{(a-b*x)})}$	0.8	0.06	11.8	11.8	0.89	10.9	12.7	35.0	5.0	0.99
Logistic	$f(x) = C/(1 + a * e^{(b*x)})$	5.3	0.09	11.1	11.1	0.74	10.3	11.8	37.8	7.8	0.98
Reciprocal	$f(x) = a + \frac{b}{x}$	9.0	-26.81	NA	9.0	0.86	8.1	9.8	40.7	10.7	0.75

Table 3: Model parameters resulting from fitting coefficients of variation (CV) of home range size of northern bobwhite coveys to the number of locations using 4 separate function models. Home ranges were calculated using the 95% fixed kernel least squares cross validation (LSCV) smoothing parameter and minimum convex polygon home range estimators, Jim Hogg County, Texas, USA, Sep-Nov, 2001-2003.

Year *Estimator*		Model parameters			Asymptote						
Model	Function	a	b	C	Estimate (ha)	SE	-1SE	$+1$SE	AIC_C	ΔAIC_C	R-square
2001 *Fixed-Kernel*											
Reciprocal	$f(x) = a + \frac{b}{x}$	0.3	3.05	NA	0.3	0.05	0.25	0.34	-0.1	0	0.93
Exponential	$f(x) = C + a * e^{(-b*x)}$	1.43	0.15	0.34	0.34	0.03	0.32	0.37	4.8	4.9	0.98
Logistic	$f(x) = C/(1 + a * e^{(b*x)})$	-0.88	-0.04	0.28	0.28	0.06	0.22	0.34	8.8	8.9	0.97
Gompertz	$f(x) = C * e^{(-e^{(a-b*x)})}$	17.67	1.82	0.5	0.5	0.04	0.46	0.53	19.4	19.5	0.84
MCP											
Reciprocal	$f(x) = a + \frac{b}{x}$	0.52	0.71	NA	0.52	0.08	0.43	0.6	8.1	0	0.58
Gompertz	$f(x) = C * e^{(-e^{(a-b*x)})}$	6.57	0.72	0.53	0.53	0.02	0.5	0.55	13.4	5.3	0.88
Exponential	$f(x) = C + a * e^{(-b*x)}$	0.76	0.07	0.43	0.43	0.13	0.3	0.56	17.3	9.2	0.79
Logistic	$f(x) = C/(1 + a * e^{(b*x)})$	-0.79	-0.01	0.24	0.24	0.4	-0.16	0.64	17.6	9.5	0.78
2002 *Fixed-Kernel*											
Gompertz	$f(x) = C * e^{(-e^{(a-b*x)})}$	2.52	0.31	0.39	0.39	0.01	0.39	0.4	-11.6	0	0.99
Reciprocal	$f(x) = a + \frac{b}{x}$	0.39	1.31	NA	0.39	0.03	0.36	0.42	-5.7	5.9	0.85
Exponential	$f(x) = C + a * e^{(-b*x)}$	0.58	0.12	0.39	0.39	0.02	0.37	0.41	-0.3	11.3	0.96
Logistic	$f(x) = C/(1 + a * e^{(b*x)})$	-0.64	-0.06	0.37	0.37	0.03	0.34	0.41	1.8	13.4	0.94
MCP											
Reciprocal	$f(x) = a + \frac{b}{x}$	0.22	2.29	NA	0.22	0.02	0.2	0.25	-10.0	0	0.97
Logistic	$f(x) = C/(1 + a * e^{(b*x)})$	-1.15	-0.17	0.31	0.31	0.01	0.3	0.32	-9.0	1	0.99
Exponential	$f(x) = C + a * e^{(-b*x)}$	2.16	0.37	0.32	0.32	0.01	0.31	0.33	-4.2	5.8	0.99
Gompertz	$f(x) = C * e^{(-e^{(a-b*x)})}$	17.02	3.02	0.41	0.41	0.04	0.37	0.45	16.9	26.9	0.80

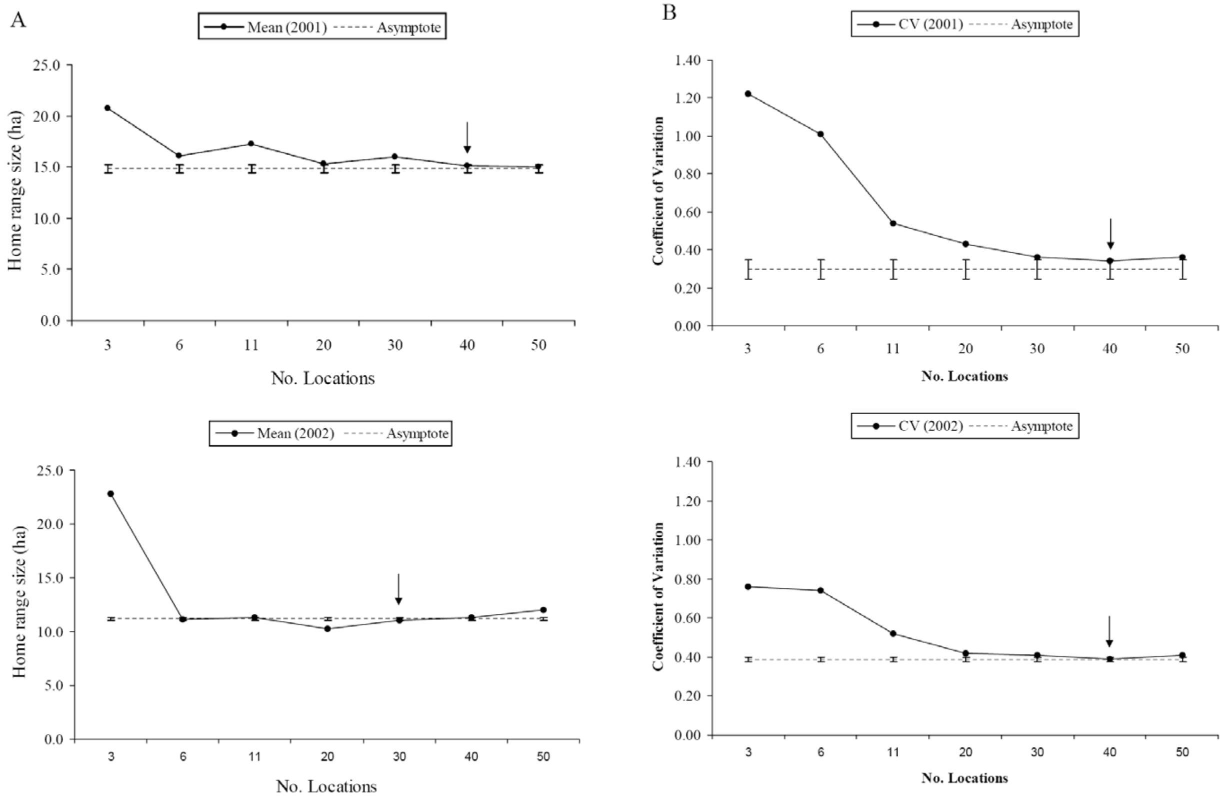

Figure 1: Asymptotes for A) mean home range size of northern bobwhite coveys calculated using 95% fixed kernel (n = 14 coveys in 2001 and n = 20 coveys in 2002) and B) coefficients of variation (CV). Asymptotes were determined by modeling mean home range size or CV as exponential, Gompertz, logistic, and reciprocal functions of the number of locations (no. locations) and then identifying the best model based on an information-theoretic framework (AIC_C). Arrows denote first observed value to fall within 1 standard error of the estimated asymptote.

els (SAS Institute, Inc. 2002-2004).

We used the asymptote obtained for the best model to estimate the minimum number of locations necessary to obtain a representative home range size for each home range estimator by year. We defined this to be the minimum number of locations when an observed point first fell within ±1 standard error of the estimated asymptote.

Results

We monitored 14 coveys in 2001 and 20 coveys in 2002 (Table 1) with an average of 2 to 3 birds in a covey. All function models provided a relatively good fit ($0.58 \leq \mathrm{R}^2 \leq 0.99$; $P < 0.05$) for area curves and CV curves for both home range estimators (Table 2, 3).

Using the 95% fixed kernel estimator, AICC scores were the lowest for the reciprocal model in 2001 with an asymptote estimate of 14.8 ± 0.38 (ha) and scores were lowest for the exponential model in 2002 with an asymptote estimate of 11.2 ± 0.12 (ha) for mean home range size (Table 2). Based on these estimates we determined that ≥40 locations were required to estimate home range size in 2001, whereas ≥30 locations were sufficient in 2002 (Figure 1). For the CV, AICC scores were lowest for the reciprocal model in 2001 with an asymptote estimate of 0.30 ± 0.05 and scores were lowest for the Gompertz model in 2002 with an asymptote 0.39 ± 0.01 (Table 3). Based on these estimates we determined

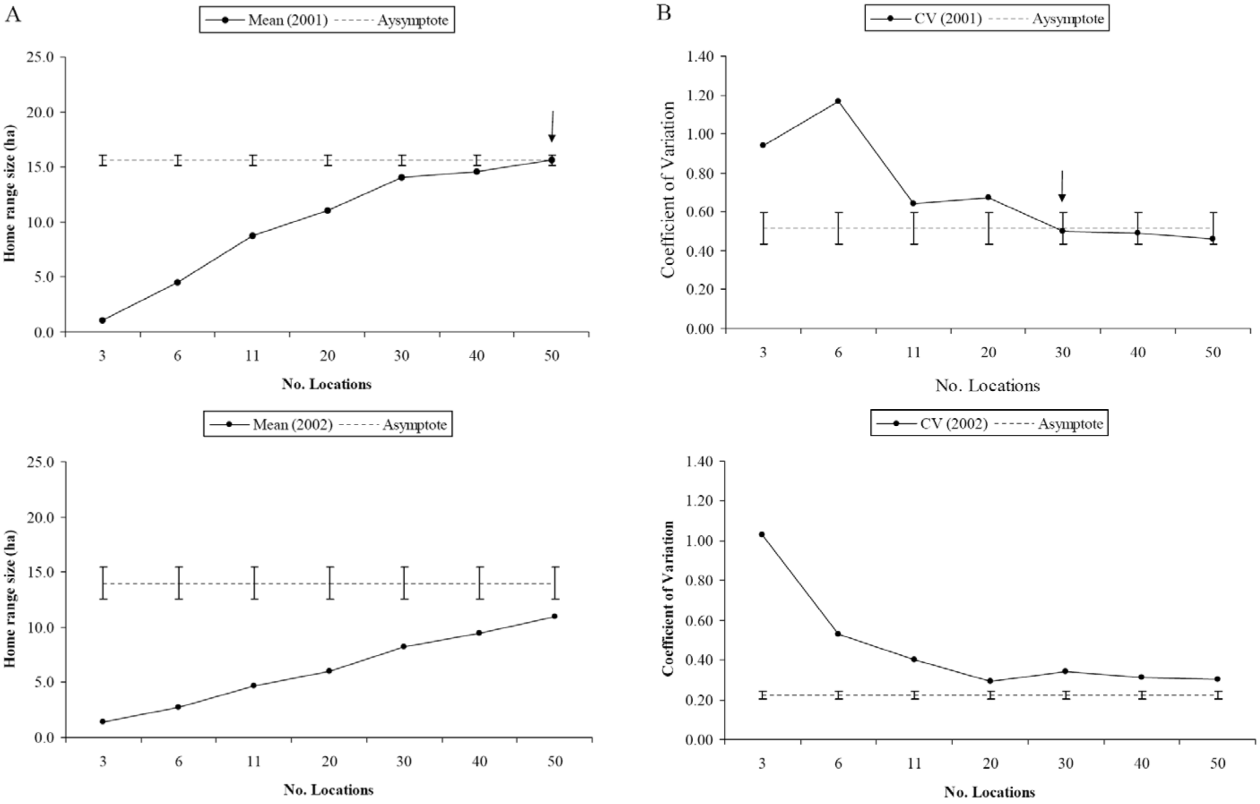

Figure 2: Asymptotes for A) mean home range size of northern bobwhite coveys calculated using minimum convex polygon (n = 14 coveys in 2001 and n = 20 coveys in 2002) and B) coefficients of variation (CV). Asymptotes were determined by modeling mean home range size or CV as exponential, Gompertz, logistic, and reciprocal functions of the number of locations (no. locations) and then identifying the best model based on an information-theoretic framework (AIC_C). Arrows denote first observed value to fall within 1 standard error of the estimated asymptote.

that ≥40 locations were required to minimize variation in home range estimation in both 2001 and 2002 (Figure 1).

Using minimum convex polygon, AICC scores in 2001 were lowest for the exponential model with an asymptote estimate of 15.6 ± 0.46 (ha) for mean home range size and AICC scores were lowest for the reciprocal model with an asymptote estimate of 0.52 ± 0.08 for the CV in 2001 (Table 2, 3). Based on these estimates we determined that ≥50 locations were required to estimate mean home range size while ≥30 locations were required to minimize variation in home range estimation (Figure 2). The AICC scores in 2002 were lowest for the exponential model with an asymptote estimate of 14.0 ± 1.46 (ha) for mean home range size and scores were lowest for the reciprocal model with an asymptote estimate of 0.22 ± 0.02 for the CV (Table 2, 3). Based on these estimates we determined that an asymptote could not be reached because actual home range size and the CV did not come within ±1 SE of the estimated asymptote calculated by the models selected by the AICC (Figure 2). Thus, there were not enough locations to estimate home range size using minimum convex polygon in 2002.

Discussion

Based on our modeling simulations we found that ≥40 locations were adequate to reach an asymptote for home range area estimation using the 95%

fixed kernel estimator for our sample of bobwhite coveys during the fall season. Our estimate using field data is similar to Seaman et al. (1999) who reported that bias and variance for the kernel estimator approached an asymptote at 50 locations using computer simulation points. They recommended using a minimum $\geq$30 locations to obtain home range estimates when using kernel estimators with LSCV, but preferably $\geq$50.

Regarding the minimum convex polygon, we documented that in 2001 $\geq$50 locations were necessary to obtain a representative home range estimate for our sample of bobwhite coveys. However, in 2002 an area-curve asymptote was not reached to obtain a representative home range. Home range estimates from the minimum convex polygon estimators continued to increase with increasing locations (a property of this estimator), though this increase was minimal in 2001. However, CV's remained relatively constant. This observation can occur because CV's are a ratio of mean:standard deviation. Therefore, similar CV's can result in spite of increasing means if their corresponding standard deviations also increase in similar proportion. Previous research has suggested a much larger number of locations (100-200) to estimate home range size using the minimum convex polygon (Bekoff and Mech 1984, Laundre and Keller 1981, Harris et al. 1990). Gautestad and Mysterud (1995) believed that asymptotes using the minimal convex polygon method would only occur when using more than several thousand locations.

Kernohan et al. (2001) evaluated 12 home range estimators, including the estimators used in this study. Overall, Kernohan et al. (2001) favored the kernel home range estimator because it required a reasonable sample size ($\geq$50 location points), had the ability to compute home range boundaries that included multiple centers of activity, was based on complete utilization distribution, was a nonparametric methodology, and lacked sensitivity to outliers. However, kernel estimators have no real comparability to other home range estimators due to its estimate being greatly affected by bandwidth choice. Minimum convex polygon also is a nonparametric home range estimator, but unlike the kernel estimator it is not impacted by bandwidth choice and can be compared to other estimators. However, the minimum convex polygon estimator requires a large sample size (i.e., $>$100 locations total), does not use utilization distribution, does not account for outliers, and does not calculate multiple centers of activity (Kernohan et al. 2001, p. 140).

Regardless of the estimator used, we recommend that verification is needed showing that an area-curve asymptote had been reached prior to home range estimation. However, identifying the asymptotes for home-range area curves has been difficult because it generally has involved much subjectivity. Previous studies identified asymptotes through visual inspection (e.g., Bond et al. 2001) or when additional locations produced $<$1% change in mean home range size (Odum and Kuenzler 1955). We estimated asymptotes by modeling mean home range or CV as a model function of number of locations. We identified the minimum number of locations when the first point fell within $\pm$1 SE of the estimated asymptote. We found that function models provided a relatively good fit for our data ($0.58 \leq R^2 \leq 0.99$) and provided a structured and useful approach for calculating area-curve asymptotes. Therefore, we recommend fitting mean home range size and CV to the number of locations using function models and an AICC score to select the best model in identifying area-curve asymptotes.

This manuscript presents a robust quantitative approach to calculating area-curve asymptotes. However, we recommend that this method be used to validate estimates of area-curve asymptotes that are based on visual inspection or the point at which there is a $<$1% change in mean home range size (Odum and Kuenzler 1955). In addition, we recommend more research be conducted to validate the strength of this method.

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Total Body Electrical Conductivity for Determining Carcass Fat in Ruffed Grouse

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Percent carcass fat is often considered a primary condition indice in game bird species. Although regarded as the standard for determining fat reserves, traditional sampling methods require sacrificing animals for chemical analysis via fat extraction. Lethal methods negate the ability to track condition of individuals through time. Avian physiology studies often require the assessment of conditional changes through time and among various treatments, which necessitate the use of a non-lethal method for estimating fat levels. We were able to accurately estimate fat condition in captive ruffed grouse (*Bonasa umbellus*) utilizing total body electrical conductivity (TOBEC). We developed predictive models to estimate percent carcass fat directly from first-order regression of TOBEC and body mass values. Validation of our best model from an independent sample (*n* = 10 individuals) produced an $R^2 = 0.85 (P < 0.001)$ for determining percent carcass fat and $R^2 = 0.89 (P < 0.001)$ for determining total fat mass in ruffed grouse. Future studies investigating galliform ecology or physiology could benefit from use of TOBEC for assessment of fat condition if non-lethal sampling is desired to track changes through time.

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Key words: *Bonasa umbellus*, carcass fat, fat condition, ruffed grouse, TOBEC, total body electrical conductivity

Introduction

Investigations of nutrition often necessitate repeated measures of individual body condition. The standard method of determining fat via proximate analysis requires the death of the animal (Reynolds and Kunz 2001). Although accurate and precise, use of this method precludes repeated measures of individual fat condition through time. Few non-lethal methods are available to estimate fat condition in gamebird species. Morphometric and body size indices have been widely used but are often inconsistent, observer-biased, and generally lack the precision of other non-lethal methods (Hayes and Shonkwiler 2001, Servello et al. 2005). Isotope dilution methods accurately estimated fat condition in chukar (*Alectoris chukar*) and domestic chickens, but require expertise as well as expensive laboratory equipment for analysis (Speakman et al. 2001, Servello et al. 2005).

In contrast, total body electrical conductivity (TOBEC) technology is a non-lethal, accurate, and relatively simple method of determining body condition in animals given appropriate validation and if hydration status and gastrointestinal fill of subjects are accounted for (Walsberg 1988, Scott et al. 2001, Servello et al. 2005). Use of TOBEC for wildlife applications was first employed by Walsberg (1988) who determined lean body mass and lipid stores in various small mammals and passerine species. TOBEC has been used to determine body composition in northern bobwhite (*Colinus virginianus*) (Roby 1991, Frawley et al. 1999), American woodcock (*Philohela minor*) (Morton et al. 1991), and ring-necked pheasant (*Phasianus colchicus*) (Purvis et al. 1999).

As part of a nutritional ecology study, we used TOBEC to estimate body composition of captive female ruffed grouse (*Bonasa umbellus*). Herein we report on the efficacy of TOBEC to accurately determine fat condition in ruffed grouse.

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Study Area

Ruffed grouse used for this research were housed in the ruffed grouse facility at West Virginia University's animal research farm in Morgantown, West Virginia, USA. A ruffed grouse colony was started in 1990 with 12 fertile eggs acquired from a wild nest found near Buckhannon, West Virginia, USA (subspecies *B. u. monticola*). Between 1991 to 2001 ruffed grouse from West Virginia, Pennsylvania, and Minnesota were added to the colony to increase genetic diversity and limit the amount of genetic crossing. At the time of this research there were 190 adult ruffed grouse at the facility. All birds are kept in individual 60×60×60-cm cages with wire floors in a curtain-sided, poultry-style building. Forced-air heaters are used in winter to keep temperatures above 10° C. All ruffed grouse are kept on a natural lighting schedule and are fed a standard turkey maintenance ration, with grit and water provided ad libitum.

Methods

Sixteen female ruffed grouse were randomly sampled to develop fat condition predictive models; 7 juvenile (<1 year) females sampled in December 2003 and 9 adult females sampled in October 2005. We sampled only females because the objectives of the overall research project focused on female ruffed grouse reproductive success in relation to body condition (A. B. Proctor and J. W. Edwards, West Virginia University, unpublished data). Prior to sampling, all individuals were assumed to be in good health given normal activity and feeding behavior, had access to free water, and were maintained on a commercial turkey maintenance ration with grit ad libitum, hence all grouse sampled were assumed to be under normal gastrointestinal fill and hydration.

TOBEC sampling

We fashioned a TOBEC scanning restraint by cutting a 53×35-cm piece of soft, pliable, opaque plastic sheeting that would extend from the tail to >2.5 cm beyond the head of an adult grouse. We used 2 sets of self-adhering Velcro strips to close the restraint. We tested the dielectric properties of the empty restraint within the TOBEC scanning chamber and found it to not register a value, indicating it would not influence sampling results. For sampling, we first tared the weight of the empty restraint on an electronic balance. We then positioned the grouse dorsally onto the open restraint and held its wings folded to the body while we snugly "rolled up" the restraint and secured the Velcro strips, making sure that legs were extended posteriorly and not positioned ventrally. Grouse appeared calm once in the restraint. It was important to secure the grouse within the restraint to restrict movement and insure that they remained motionless during the TOBEC scanning process (EM-SCAN Inc 1993).

We weighed each grouse to the nearest 0.1 g on an electronic balance prior to determining a TOBEC value using an EM-SCAN Model SA-3000 small animal body composition analyzer with a 114 mm Model 3114 detection chamber (EM-SCAN, Springfield, Illinois, USA). We recorded 5 scans to obtain an average TOBEC value for each grouse. Total sampling time (mass determination, placement in restraint, and 5 TOBEC scans) averaged 8-10 min. EM-SCAN Inc (1993) recommends that the coefficient of variation of all measurements for individual subjects not exceed 3%. In preliminary trials, we found that a 3% coefficient of variation approximated a 20-unit range among 5 scans. Therefore, we would record 5 scans initially; if the range of these scans exceeded 20 units, outliers were discarded and additional scans were taken until the 3% coefficient of variation requirement was satisfied (Frawley et al. 1999, Purvis et al. 1999). Immediately following TOBEC sampling, we sacrificed grouse via carbon dioxide asphyxiation. Handling and euthanasia procedures followed West Virginia University's Animal Care and Use Committee protocol number 03-0913. Sacrificed grouse carcasses were placed in air-tight plastic bags and frozen.

Proximate analysis

Carcasses were allowed to partially thaw and prepared by removing feathers, head, legs below the tibio-tarsus-tarsometatarsus junction, and gastrointestinal and reproductive tracts (Norman and Kirkpatrick 1984). The remaining carcass was cut into 2-3 cm pieces, ground in a commercial meat grinder and frozen. Frozen ground contents were lyophilized to constant mass to determine moisture content. Lyophilized contents were homogenized in a commercial blender and subsampled for analysis. Proximate analyses of samples were performed in duplicate at West Virginia University's Rumen Fermentation Profiling laboratory. Percent fat of sacrifice homogenates was determined using ether extraction in a Sohxlet apparatus following the Association of Official Analytical Chemists (AOAC) protocol 920.39 (Association of Official Analytical Chemists 1990).

Statistical Procedures

Percent carcass fat values were arcsine transformed (Zar 1999) and tested for normality (PROC UNIVARIATE, SAS Institute, Inc. 2002-2004). A first-order polynomial regression model was expected to best explain predicted total fat mass and percent carcass fat from chemical analysis of the 16 grouse used for predictive models (Scott et al. 2001). We developed *a priori* candidate models for total fat mass and percent carcass fat using body mass and TOBEC value as predictor variables. We used a global model incorporating both body mass and TOBEC value (models 3 and 6) to explain percent carcass fat and total fat mass, as well as each predictor variable on its own (models 1, 2, 4, and 5). We used regression analysis (PROC REG, SAS Institute, Inc. 2002-2004) to develop predictive models for total fat mass and percent carcass fat. We used direct models to predict total fat mass and percent carcass fat from TOBEC value and body mass. Morton et al. (1991) and Snyder et al. (2005) recommended the use of direct models for predicting fat over 2-stage models where predicted lean mass is subtracted from total body mass due to increased relative error associated with the latter approach. Three candidate models were used to predict total fat mass (TFM) and percent carcass fat (PCF), respectively:

Model 1: TFM = Body mass

Model 2: TFM = TOBEC value

Model 3: TFM = Body mass + TOBEC value

Model 4: PCF = Body mass

Model 5: PCF = TOBEC value

Model 6: PCF = Body mass + TOBEC value

We evaluated models based on Akaike's Information Criterion (AIC) adjusted for small sample size (AIC_c), AIC_c differences (Δ_i), and Akaike weights (ω_i) (Burnham and Anderson 2002). Models with AIC_c differences ≤ 2 were considered competing models (Burnham and Anderson 2002). Akaike weight (ω_i) estimates the probability that a particular model is the best model in the candidate set (Burnham and Anderson 2002).

We validated our best models on an independent group of 10 female ruffed grouse (5 juveniles and 5 adults) that were sampled for body mass and TOBEC value and sacrificed on 18 February 2005. This validation set was sampled and processed exactly as the 16 grouse used in model development. Statistics are reported on transformed data in this manuscript while results are shown for untransformed data.

Results

Mean coefficient of variation of TOBEC values among all sacrifices was 1.08%. Total body water of 16 grouse used for predictive equation formation was 77.45 ± 0.64% (mean ± SE, range = 72.70-81.41), and 75.15 ± 1.13% (70.44-83.44) for grouse from the validation group. Percent carcass fat of grouse used in predictive models was 15.97 ± 2.19 (3.38-30.89), and 23.15 ± 3.27 (2.85-37.31) for those in the validation set. Our global models were the only supported models for predicting TFM (Model 3, ω_i = 0.98) and PCF (Model 6, ω_i = 0.98) (Table 1). In both best models, live body mass was positively related to TFM and PCF:

Model 3: TFM = $-79.457 + (0.310 \times \mathrm{BM}) - (0.164 \times \mathrm{TOBEC})$

Model 6: PCF = $-27.621 + (0.155 \times \mathrm{BM}) - (0.082 \times \mathrm{TOBEC})$

Table 1: Information theoretic model selection using Akaike's Second Order Criterion (AIC$_c$) for determining predicted total fat mass (TFM) and percent carcass fat (PCF) for both years (n = 16). Data fit using logistic regression in SAS (PROC REG, SAS Institute 2002-2004).

Model	K	Log-L^a	AIC_c	Δ_i	ω_i	R^2
Total fat mass						
TFM = Body mass + TOBEC	4	-28.24	68.12	0.00	0.98	0.83
TFM = Body mass	3	-34.11	76.21	8.09	0.02	0.64
TFM = TOBEC	3	-41.33	90.67	22.55	0.00	0.11
Percent carcass fat						
PCF = Body mass + TOBEC	4	-16.95	45.54	0.00	0.98	0.83
PCF = Body mass	3	-22.85	53.70	8.15	0.02	0.65
PCF = TOBEC	3	-30.19	68.38	22.84	0.00	0.12

^{a}Log-likelihood value

Where,

BM = Live body mass (g)

TOBEC = Average value of 5 TOBEC scans

Body mass and TOBEC were excellent predictors of fat condition in our validation samples explaining 85% ($R^2 = 0.85, P = 0.001$) of variation in percent carcass fat and 89% ($R^2 = 0.89, P = 0.001$) of variation in total fat mass among individuals. Our relative error for predicting percent carcass fat was 3.73 ± 1.62% and our absolute error was 7.62 ± 4.94 g (mean ± 95% confidence interval) (Table 2).

Discussion

We found measures of TOBEC and body mass to accurately predict carcass fat in ruffed grouse. Moreover, the addition of TOBEC as a response variable in our models substantially increased the amount of variation in carcass fat explained compared to models where body mass was the single predictor variable. Previous studies of galliform fat condition using TOBEC have reported mixed results. Roby (1991) reported TOBEC as a reliable estimator of fat condition from a sample of 52 captive and 11 wild-caught bobwhite quail, explaining 92% of the variation in total body lipid. Frawley et al. (1999) found body mass to be the best predictor of fat condition in bobwhite quail and reported limited predictive support when combining TOBEC and body mass measures. Purvis et al. (1999) reported that fat estimation from TOBEC and body mass in wild ring-necked pheasant was highly variable, and suspected that the variation in precision was due to hydration status and gastrointestinal fill. Consistent hydration status and normal feeding are important considerations to accurately assessing fat condition. Our use of captive birds with access to free water and feed likely reduced variation in these factors that might be experienced in wild populations.

Use of TOBEC for body condition studies requires a brief acclimatization period where one must become comfortable with the device and method of subject restraint. Throughout our trials, the TOBEC unit would occasionally produce obviously erroneous scan outputs. For example, where previous scans on a subject centered around 400 (TOBEC units), the next might have been 1200, which was obviously an erroneous scan and should be recorded as such. We suspected that electrical fields within the facility could have produced these results but were never certain. EM-SCAN Inc (1993) cautions that

Table 2: Validation of best models for fat mass (g) and percent carcass fat on an independent sample of female ruffed grouse (n = 10) from West Virginia University's Animal Sciences Farm, Morgantown, West Virginia, USA, 2003-2005. Differences between actual and predicted values are absolute. Body mass (g) values were recorded directly before final TOBEC sampling and sacrifice.

Age class[a]	Live mass	Actual fat mass	Predicted fat mass	Fat mass difference[c]	Actual % fat	Predicted % fat	% fat difference[d]
A	530.2	34.22	31.61	2.60	25.66	22.74	2.92
A	406.7	1.92	3.65	1.73	2.85	6.02	3.17
A	661.1	70.99	48.64	22.35	36.32	32.41	3.91
A	616.4	36.27	39.08	2.81	25.88	26.72	0.84
A	573.6	60.95	39.19	21.75	37.31	27.18	10.13
J	611.8	50.55	40.45	10.10	30.59	27.63	2.96
J	490.1	21.39	15.45	5.94	18.42	12.79	5.63
J	431.0	16.67	12.79	3.88	15.54	11.63	3.91
J	554.1	12.74	24.70	3.04	19.74	18.11	1.63
J	510.2	24.58	22.36	2.21	19.21	17.02	2.19
S.E.	25.8	6.66	4.55	2.52	3.27	2.68	0.89
Mean	538.5	34.53	27.79	7.62	23.15	20.22	3.73
95% CI[b]	±0.6	±13.04	±8.92	±4.94	±6.41	±5.25	±1.62

[a] A = adult, J = juvenile

[b] 95% confidence interval

[c] Fat mass difference = predicted fat mass − actual fat mass

[d] % fat difference = predicted % fat − actual % fat

the area chosen to use a TOBEC unit should be as far away as possible from electrical equipment and other possible sources of electrical fields. We found these erroneous scans to occur at a low rate and not effect our overall ability to arrive at an acceptable TOBEC value.

Our findings support the use of TOBEC to assess fat condition in ruffed grouse in captive studies. If hydration state and nutritional conditions are monitored and calibration procedures followed, it can provide a relatively simple method to accurately determine fat condition. Moreover, because it is nonlethal, it is possible to determine repeat measures of fat condition on individuals over time.

Management Implications

Many wildlife nutritional and conditional investigations require that captive animals be used to best represent conditions present in wild populations. In such studies where repeated measures of body condition are necessary, TOBEC can be a viable means to accurately determine percent fat (and hence lean mass) of animals if proper calibration procedures are used. We found the use of TOBEC to be a quick and easy method to determine condition of female ruffed grouse and that the different sizes of EM-SCAN scanning chambers would facilitate its use across a wide variety of galliform species.

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Age Determination of Pheasants (*Phasianus Colchicus*) using Discriminant Analysis

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Most gallinaceous birds can be identified as juveniles or adults using the outermost primaries (P9 and P10) which are retained until after the first breeding season and are often identifiable by colour and wear. The pheasant *Phasianus colchicus*, however, moults all ten primary feathers during its post-juvenile moult so alternative techniques are required. To date the method most widely used on live birds is measurement of the shaft diameter of the proximal primary feather, P1, which is replaced first before the bird is fully-grown. Using a known-age sample of 752 free-living pheasants, this study presents a discriminant function analysis using proximal primary feather measurements and other morphological characteristics to achieve a greater level of accuracy of ageing. Ageing accuracy was high, especially for males, at over 95%. The model was less accurate for females, with 83% and 94% respectively for the two year groups. When our model was applied to an independent data set of unknown-age birds 85% were classified. Less than 3% could not be aged accurately and the remainder were unclassified due to missing measurements. Our model offers a reliable method of ageing pheasants, both live and dead, however researchers are cautioned to potential year, origin (stock) and site effects.

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Key words: ageing, discriminant function analysis, *Phasianus colchicus*, proximal primary feather, pheasant

Introduction

The ability to age pheasants is valuable in population dynamics studies because age affects many biological parameters, including survival probability, breeding status and reproductive success (Brittas et al. 1992, Woodburn 1999). It can also be useful to know the age structure of pheasant populations in field experiments so that the effect of age can be taken into account. In general ageing techniques classify birds into 2 groups rather than into specific year classes (Wishart 1969, Sayler 1995, Newton 1998). Pheasants are generally classed as juvenile if they are <1 year old (birds entering their first spring), and adults thereafter.

For many gamebirds plumage characteristics provide the most reliable means of separating juveniles from adults. In most species primary flight feathers are moulted sequentially, starting with the proximal (innermost) feather, and progressing distally in a fairly regular time pattern. Typically primaries P9 and P10 are retained until after the first breeding season, providing a means of ageing. In juveniles they may be more worn, duller in colour or shaped differently compared to adults (Dimmick and Pelton 1994).

Pheasants differ in that they moult all 10 primaries during their post-juvenile moult rather than retaining P9 and P10 (Petrides 1942), which makes ageing by feathers alone more difficult. Game biologists have tried several techniques for ageing pheasants, with varying degrees of accuracy. Some of these are:

- Bone histology involves examining the layered appearance of very thin sections of bone from pheasant legs. It is a successful technique for ageing males, but because of resorption of

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bone to supply calcium for egg shells it is unreliable for females (Stone and Morris 1981). Another disadvantage is that it can only be applied to dead birds.

- A jaw test is sometimes used by hunters in the field (Linduska 1943). The force required to break the lower jaw is less in juveniles because of the incomplete calcification of their bones. This method is not accurate enough on its own (Nelson 1948), and cannot be used on live birds.
- Eye lens weight has been used with reasonable success in some bird species (Payne 1961, Campbell and Tomlinson 1962), but has not been found to be useful for pheasants since it can only separate adults and juveniles reliably in autumn (Dahlgren et al. 1965) and can be used only on dead birds.
- The Bursa of Fabricius is a small sac-like cavity opening into the cloaca of birds. In juvenile pheasants the bursa is evident and usually between 15-40 mm deep, but is very shallow or completely closed in adults (Linduska 1943, Kirkpatrick 1944). The depth of the bursal cavity provides a good indicator of age during autumn and early winter, but after January it begins to regress in juveniles making this method less reliable. Although this test is easier to perform on dead birds it can be used on live birds as well, but it may be quite stressful.
- Ageing by measuring spur length is applicable only to males. It is fairly reliable until December, because after December worn-down spurs of older males and the growing spurs of young males can overlap in length making age determination based on spur length alone unreliable (Linduska 1943, Stokes 1957, Gates 1966).
- Primary shaft diameter involves measuring the diameter of the shaft of the proximal (innermost) primary; the first primary shed during the post-juvenile moult. Because it is replaced before the bird is fully grown and retained until moulting the following year (Westerskov 1957), the proximal primary of a fully-grown juvenile is likely to be smaller than that of an adult (Wishart 1969). Using this method, Greenberg et al. (1972) found that the separation between the two age classes was 98% reliable in males and 92% reliable in females, while Robertson (1985) found 100% and 83% respectively. The method can be applied to both live and dead birds and used throughout the year, but requires calibration for each pheasant population examined.

In this study we applied discriminant function analysis to age a sample of pheasants. This technique has been widely used in biological studies of many species to differentiate between groups. In insect systematics it has been used for groups of closely-related species that are morphologically very similar (Barker 1998) and where environmental variation within species may mask between-species differences (Blackman 1992). It has also been valuable in sexing birds, which are sexually monomorphic in plumage (Kavanagh 1988, Green and Theobald 1989, Clark et al. 1991).

The objective of this investigation was to determine whether discriminant function analysis using a combination of morphological parameters together with proximal primary feather measurements from known-age tagged pheasants could be used to age untagged birds from the same population more accurately than just using feather data alone.

Study Area

The study was carried out on an area of predominantly arable farmland in Dorset, southern England (Grid Reference SU 0119). It covers an area of 400 hectares, with 10% of the area being broadleaved woodland and 3% permanent grassland. Hand-reared pheasants (reared intensively in pens) were released on the study area each year to supplement the population for shooting during the winter. All birds were tagged with individually numbered patagial wing-tags at the time of release in late summer. A proportion of the spring breeding population successfully reproduced in the wild each year, as determined by annual brood counts after harvest, so the resident pheasant population was a mixture of hand-reared birds and parent-reared offspring of previously hand-reared birds.

Methods

Data were collected from pheasants during February and March between 1988 and 1995, when a proportion of the birds were caught in walk-in funnel catchers (Woodburn 1999). Several body measurements were recorded from each bird including body weight, tarsus length, head length and spur length in males. A proximal primary feather (innermost) was removed and if the bird had not been previously released (and therefore tagged) then it was also tagged with an individually numbered patagial wing-tag.

The proximal primary feathers collected in spring were placed in a drying oven at 50° C for 24 hours before being measured (within 8 hours of drying). This helped to reduce variation in the measurements due to relative humidity (Greenberg et al. 1972). The shaft diameter was measured at the level of the cuticle tissue scar near the base of the barbs in the same plane as the vane (Wishart 1969). Measurements were taken to the nearest 0.02 mm by sliding the feather into a tapering aperture varying from 1.5 mm to 4.5 mm, as described by Robertson (1985). Feather lengths were also measured to the nearest 0.1 mm when the feather was flattened and straightened.

Using measurements of body weight, tarsus length, head length, spur length, ratios of body weight to tarsus length and head length and proximal primary feather diameters and lengths, statistical comparisons of means of groups of known-age individuals were made based on the student's t-test. Subsequently, these data were used in a discriminant function analysis (Sokal and Rohlf 1981, Green 1982). A discriminant function analysis seeks the single linear combination of all or some of the measured variables that best discriminates between groups. The function can assign a probability of an individual being in each group (Green and Theobald 1989). Unknown individuals can then be assigned to previously defined groups.

A series of multivariate discriminant function analyses were determined using SYSTAT (Wilkinson 1990). Feather variables and all morphological variables were examined and reduced through a forward stepwise procedure to achieve the smallest subset of predictors that correctly classified the maximum number of individuals.

We used 988 pheasants in the analysis. Of these 752 were of known age and 236 were of unknown age. Data were analysed as two separate groups for both sexes because in 1988-1990 neither head length nor spur length in males were measured. The birds were split into 2 groups based on the year they were caught: -

Group 1 - pheasants caught in 1988-1990

Group 2 - pheasants caught in 1991-1995

Before doing this analysis data from the known-age birds was randomly split such that two-thirds (503 birds) were assigned to a predict group and one-third (249 birds) to a test group. The predict group was used to compute the discriminant function and the test group was used to cross-validate the function using a separate group of known-age birds.

Since the data were collected over a number of different years and to account for any year effect, the forward stepwise procedure was initially run without the year variable. Once the predictor variables were determined the stepwise was then re-run adding in the year variable. This enabled us to assess whether adding year made a significant improvement to the prediction accuracy of the model, and to determine the change in prediction accuracy.

Results

Morphological characteristics

We assessed the normality of the independent variables, grouped by sex and age. All were normally distributed except body weight in juveniles.

Table 1: Comparison of morphological variables and feather measurements for male and females adult and juvenile pheasants collected during spring 1988-1995, Dorset, southern England. The last column shows the success rate of prediction accuracy of age from a discriminant function analysis upon each individual variable alone.

	Adults		Juveniles			
	n	Mean ± SD	n	Mean ± SD	P^a	Level of prediction accuracy
Males						
Body weight	46	1581.63 ± 102.43	254	1459.65 ± 123.48	***	70%
Tarsus length	49	72.5 ± 2.36	256	71.7 ± 2.85	n.s.	56%
Head length	42	77.28 ± 2.59	169	75.79 ± 2.33	***	62%
Primary shaft diameter	46	3.62 ± 0.16	240	3.21 ± 0.19	***	87%
Primary shaft length	46	17.91 ± 0.57	197	16.48 ± 0.66	***	88%
Maximum spur length	41	23.53 ± 1.99	168	20.25 ± 1.76	***	82%
Ratio of body weight to tarsus length	46	21.86 ±1.30	252	20.37 ± 1.52	***	70%
Ratio of body weight to head length	39	20.78 ± 1.34	167	19.27 ± 1.55	***	69%
Females						
Body weight	85	1198.06 ± 99.68	331	1137.36 ± 106.14	***	62%
Tarsus length	87	63.94 ± 2.72	333	63.34 ± 2.7	n.s.	55%
Head length	73	68.56 ± 1.92	306	67.92 ± 2.3	*	58%
Primary shaft diameter	80	3.33 ± 0.17	315	2.86 ± 0.19	***	89%
Primary shaft length	80	16.23 ± 0.59	265	14.89 ± 0.61	***	87%
Ratio of body weight to tarsus length	84	18.77 ± 1.47	330	17.96 ± 1.62	***	60%
Ratio of body weight to head length	71	17.56 ± 1.39	305	16.74 ± 1.44	***	63%

[a] Student's t-test ($* = P < 0.05, *** = P < 0.001$).

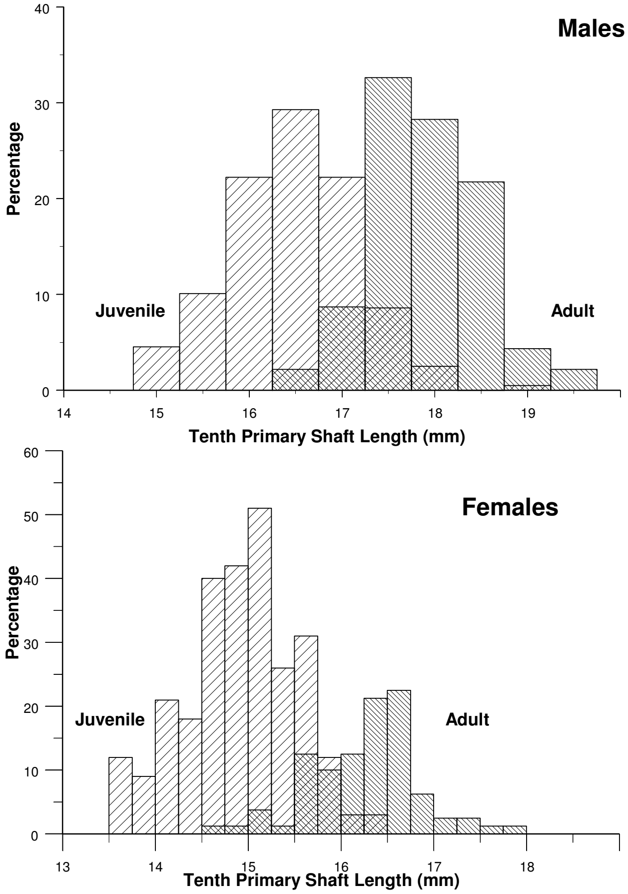

Figure 1: Frequency distribution of proximal primary feather lengths from male and female pheasants caught during spring 1988-1995, Dorset, southern England. The dark hatched area show where the adult and juvenile values overlap.

Consequently log (body weight) was used in subsequent analyses.

Comparison of means for all the morphological variables and feather measurements for each sex showed that adults had higher values than juveniles, with the exception of tarsus length in both sexes (Table1). However there was considerable overlap between adults and juveniles in some measures. The frequency histograms of the two feather measurements and spur length in males showed the smallest degree of overlap between the ages (Figs 1, 2, and 3).

Table 1 also shows the actual level of accuracy of prediction for all the morphological variables and the two feather measurements taken individually. For both sexes the feather data provided the highest level of prediction accuracy.

Multivariate discrimination

Table 2 shows the results of the discriminant function analysis showing the smallest subset of predictor variables that best discriminates between the ages. The change in prediction accuracy is shown as more variables were selected by the model. As described in Methods, some morphological variables were not measured in the early years of the study and so the data were analyzed separately as Group 1 (1988-1990) and Group 2 (1991-1995).

In all cases the feather variables were important predictors, especially proximal primary shaft diame-

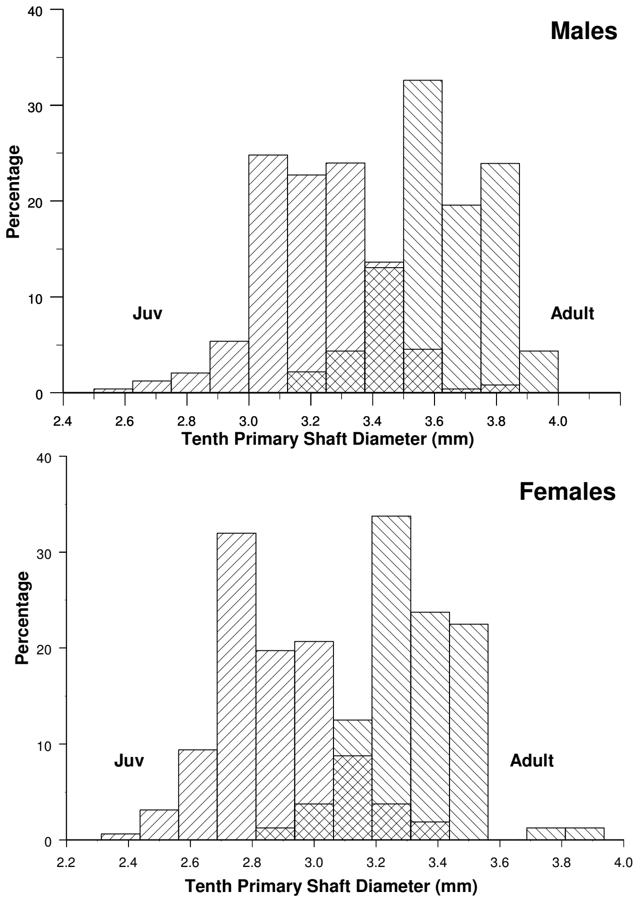

Figure 2: Frequency distribution of proximal primary feather shaft diameters from male and female pheasants caught during spring 1988-1995, Dorset, southern England. The dark hatched area show where the adult and juvenile values overlap.

ter. In males spur length, where measured, was also shown to be important.

The variable year was shown to have no effect on the accuracy of prediction in both groups of males and in Group 1 females but it was selected in the forward stepwise of Group 2 females (years 1991-1995), where it improved the accuracy of prediction by 3%. In the initial stepwise where all the measured variables were included and year was excluded, the log(body weight) variable was selected for Group 2 females. However, when the stepwise procedure was re-run using the selected variables and including year, log(body weight) was dropped but year was then selected, suggesting that the two variables are highly correlated. We examined this and found that body weight did vary between years for this group of females, ($F_4 = 4.063$, $P < 0.01$).

Cross validation

The test group of known-age birds was used in cross validation to check the accuracy of the discriminant function. Table 3 shows the classification success of both the test group and the predict group of birds from each of the two year categories for both sexes.

In all cases the classification accuracy of the test group was similar to that of the predict group used to compute the original discriminant function.

After cross validation, the discriminant functions

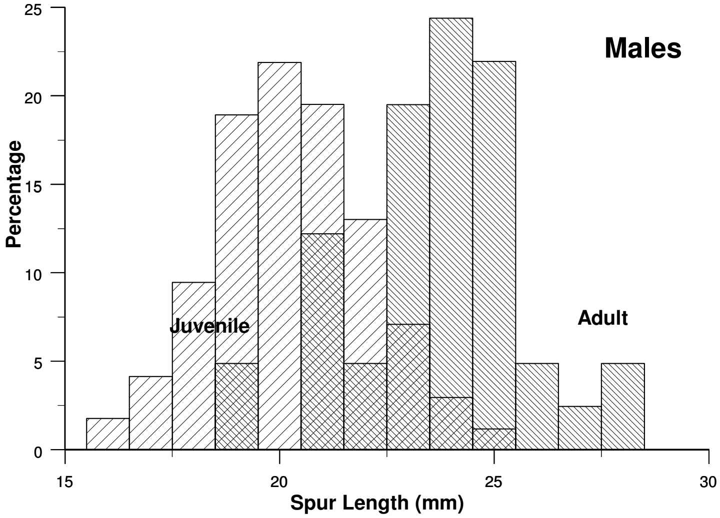

Figure 3: Frequency distribution of maximum spur length of male pheasants caught during spring 1988-1995, Dorset, southern England. The dark hatched area show where the adult and juvenile values overlap.

were then used to predict the age class of 236 previously unknown-age individuals for which feather and morphological data were collected (Table 4). The analysis assigned each bird a probability of being in each of the 2 age groups. The bird was given the age of the group with the higher probability value. As shown in the table, a small percentage of birds could not be aged because they were either borderline with almost equal probability of being assigned to the adult or juvenile group or they had missing values for some variables and their data were excluded from the analysis.

Discussion

In this study the comparison of means of the morphological variables and the two feather measurements showed there was considerable overlap between adults and juveniles. This indicated how difficult it would be to accurately age a proportion of the birds using any one variable alone. Discriminant function analysis has been shown to provide a suitable method of highlighting the key variables important in predicting the age of pheasants. From our results both feather measurements, proximal primary shaft diameter and proximal primary shaft length, were important predictor variabales, especially primary shaft diameter. This was true for both sexes but in males we found that spur length was also an important predictor variable. Including other morphological variables did not significantly improve the accuracy of ageing in either males or females.

We achieved greater accuracy of ageing in males (98%) compared to females (94%). In particular we found reduced accuracy of prediction in the Group 1 females which may partly be due to the small sample size used in the analysis. In the early years of the study we had missing values for some of the measured variables. In the analysis all data from an individual where there was not a complete set of variables was omitted. Therefore, in some cases where for example the feather length was not recorded because the feather tip was broken, all data from that individual bird was excluded from the analysis, thereby reducing the sample size.

Our findings are similar to those of Greenberg et al. (1972) who studied wild pheasants in Illinois. They assessed the use of proximal primary feather diameter and length measurements as an ageing

Table 2: Variables selected in multivariate discriminant function analysis to predict the age of a known-age sample of pheasants. Classification success shows the change in accuracy of prediction at each step in the forward stepwise model. Data were collected from pheasants caught in spring 1988-1995, Dorset, southern England.

	Step	Variables	Canonical discriminant functions	Classification success (%)
Males				
	0	constant	-32.214	
1988-1990	1	primary shaft diameter	4.996	95%
n = 63	2	primary shaft length	0.919	**98%**
	0	constant	-25.855	
1991-1995	1	primary shaft diameter	3.69	93%
n =148	2	spur length	0.215	94%
	3	primary shaft length	0.556	**95%**
Females				
1988-1990	0	constant	-23.286	
n =30	1	primary shaft diameter	7.228	**83%**
	0	constant	-25.438	
1991-1995	1	primary shaft diameter	3.644	89%
n =262	2	primary shaft length	0.925	91%
	3	year	0.283	**94%**

technique. Pheasants were captured in autumn and winter and separated into juvenile and adult age classes on the basis of bursal depths. The level of accuracy achieved by Greenberg et al. (1972) was similar to that found in this study, varying from 92-98% in males and 90-92% in females. They found that including the lengths of the proximal primaries did not improve the level of ageing accuracy and they did not include any other morphological variables in their analysis. They did not assess the age of birds beyond January-February.

The variable year did not affect the accuracy of

Table 3: Prediction success in ageing a subset of a known-age pheasants (test group) using previously defined discriminant functions derived from a separate sample of known-age pheasants (predict group). Data were collected from pheasants caught in spring 1988-1995, Dorset, southern England.

		Predict group classification success (%)	Test group classification success (%)
Males	Group 1	98%	91%
	Group 2	95%	95%
Females	Group 1	83%	92%
	Group 2	94%	96%

Table 4: Predicted age of unknown-age pheasants using discriminant function analysis. Borderline birds could not be accurately assigned an age group and unclassified birds had missing values for one of the required parameters. Data were collected from pheasants caught in spring 1988-1995, Dorset, southern England.

		Predicted Adult	Predicted Juvenile	Borderline	Unclassified
Males	Group 1 $n = 44$	14 (32%)	23 (52%)	0	7 (16%)
	Group 2 $n = 46$	6 (13%)	34 (74%)	1 (2%)	5 (11%)
Females	Group 1 $n = 35$	6 (17%)	21 (60%)	2 (6%)	6 (17%)
	Group 2 $n = 111$	19 (17%)	77 (69%)	3 (3%)	12 (11%)

prediction in either group of males or in Group 1 females but it did have an effect on the results from the Group 2 females. Body weight was initially selected as a predictor variable in this group but was dropped when year was included, suggesting high correlation between the two variables. Further analysis showed that female body weight did vary between years, particularly in juveniles. This may reflect food availability in different years and nutritional status of the females. From 1992 onwards the hand-reared pheasants on the study area were bought as six-week old poults from game farms and put directly into release pens on the farm. Prior to this the pheasants were bought as one day-old chicks hatched at the game farm from eggs collected from hens on the study area They were hand-reared in pens on the study area and released into the wild at six-weeks old. This difference in management practice between years together with variation in the genetic stock of the birds from the game farm could also contribute to the year effect shown in the females.

When applying discriminant analysis to pheasants it is important to note that birds from different areas may show regional variation in morphological characteristics. This could affect the accuracy of the ageing technique. Therefore, pheasants that are to be aged should ideally be from the same population as those birds used to determine the final discriminant equation. This was also suggested by Robertson et al. (1985). Several other studies have also found variation in mean size of primaries from different pheasant populations, and have concluded that to accurately age unknown birds, feather measurements from known-age birds from the same population should be used (Greenberg et al. 1972, Goransson 1982). As already mentioned above, there is also the potential for variation in populations as a result of different management practices. The quality and quantity of food available to birds is one factor, but differences in habitat and climate could also influence morphological variables, such as feather size.

It is also possible that variation in morphological and feather measurements may be due to the origin of the birds, and where possible, this should be taken into account. Sage et al. (2001) found body weight differences in spring between females of wild origin compared to those of hand-reared origin. In their study pheasant eggs from both a wild pheasant area and from an area populated by hand-reared pheasants were collected. The chicks were then hatched, intensively-reared and released together under identical conditions such that the only difference between the two groups of birds was their genetic origin. The scientists found that wild birds weighed less than those originating from hand-reared birds, but there were no differences in tarsus length and head length between the groups. In contrast Hill and Robertson (1988) found no difference in body weight between populations of wild and hand-reared hen pheasants measured in spring. Wishart (1969) compared measurements of shaft diameter and shaft length of proximal primaries from hatchery reared and wild pheasants from the same region. No differences were found between the groups.

It was not possible to test the effect of origin on the measured variables in the data set used in this study because the sample size of known-age juvenile and adult birds reared in the wild by their natural parent was too small. It was therefore assumed that there were no differences in either the morphological or feather data collected from the wild and hand-reared birds on the study area. The justification for this assumption is that the wild group were likely to be the offspring of previously hand-reared birds, and therefore were not genetically different from the hand-reared group itself. Also, there was a history of pheasant rearing and releasing over several decades on the land surrounding the study area and so, any truly wild birds that may once have been in the area would have undoubtedly interbred with free-living hand-reared birds.

Management Implications

The results from this study suggest that pheasants can be accurately aged using length and diameter measurements of their proximal primary feathers together with spur length measurements in males. The advantages of using this method for ageing are that large samples of data can be collected quickly and easily, no expensive equipment is needed and, unlike some methods, it can be used on live birds. Wildlife managers will also find it a valuable method as it is applicable throughout the year and not confined to autumn and early winter.

It is important, however, that researchers are aware of potential year, origin and site effects when using this technique. The pheasants used in this study were either hand-reared in origin or were the offspring of previously hand-reared birds. It is possible that different results would have been obtained if sampling from a population of wild birds reared naturally by their mother with no influence of hand-rearing. This should be taken into account, although studies by Wishart (1969) and Hill and Robertson (1988) suggested there were no differences in feather and morphological measurements of wild and hand-reared birds. However, to reduce the likelihood of these factors having an effect, when applying the model the discriminant function equation should be derived using data from a sample of known-age birds taken from the same population as those to be aged.

Being able to age birds accurately will greatly enhance our understanding of pheasant biology. From a management point of view it may be valuable to determine the ratio of old to young birds in the bag during the shooting season. More importantly being able to distinguish between first-year and older birds in spring allows the age structure of a breeding population to be established. By means of individually marking birds or using radiotelemetry detailed information can be collected on breeding behaviour, reproductive performance and survival of pheasants in relation to age (Woodburn 1999).

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Telemetry Effects

Additional Evidence Against Radio-handicapping of Northern Bobwhites

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The validity of radio-telemetry to produce reliable information (e.g., survival estimates) has recently been challenged. Radio-telemetry is a widely used technique in studies of numerous species, therefore, concerns regarding potential bias in these estimates warrant further investigation. As such, and as part of a larger study, we investigated 3 aspects of potential radio-bias: 1) variation in survival distributions among treatment (newly radio-tagged) and control (previously radio-tagged) groups; 2) proportion of trapped animals censored during the traditional 7-day censor period; and 3) ramifications to cause-specific mortality through estimation of harvest rate. Kaplan-Meier survival, based on 30-day post trapping, was similar between treatment (n = 901) and control (n = 293) bobwhites for all but 1 of 8 trapping sessions during 2000-2004. In this case, treatment bobwhites (0.970, SE = 0.015) had higher survival than control birds (0.878, SE = 0.042). We determined the effect of censoring relative to sample size was inconsequential for our analysis because the proportion of bobwhites (18 out of 1,350; 0.013) meeting the criteria for censoring, i.e., dying during the first 7 days, was minimal. Censoring of these data influenced survival estimates by an average of only 0.016 (SE = 0.004; range: 0.00 - 0.04). We evaluated harvest rate by comparing first year recovery rates of banded verses radio-tagged birds during thirteen hunting seasons occurring between 1992 and 2005. Annual recovery rate was not different ($P < 0.05$) for banded birds and radio-tagged birds where harvest averaged 6.68% (range 3.3 - 11.7) and 6.65% (range 3.4 - 11.1), respectively. These findings are consistent with previous research demonstrating that radio-telemetry can provide reliable demographic information. However, we recommend that future researchers test for these potential effects among their data before making biological inferences.

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Introduction

Over the last two decades, a large amount of demographic and behavioral information has been generated from the use of radio-telemetry on northern bobwhite (hereafter bobwhite; *Colinus virginianus*). The reliability of this information depends on meeting the primary assumption that marking individuals with radio-transmitters does not handicap them in any way and that these individuals are representative of the population at large (Pollock et al. 1989). Recent articles have questioned these underlying assumptions, suggesting that researchers are "radio-handicapping" bobwhites and should therefore, be skeptical of information generated from these studies (Parry et al. 1997, Cox et al. 2004, Guthery and Lusk 2004). More recent empirical analyses from large-sample and long-term studies by researchers in the southeastern U.S. have addressed these criticisms. In particular, Palmer and Wellendorf (2007) and Terhune et al. (2007) compared survival rates for banded versus radio-tagged birds based on mark-recapture and recovery analysis from a large sample of bobwhites. They found no difference in survival rates between groups and showed that their estimated rates were similar to those derived simultaneously from radio-telemetry. Additionally, Sisson et al. (2009) countered Guthery and Lusk's (2004) argument that telemetry based survival estimates are biased low by presenting

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49 radio-telemetry generated survival estimates for their respective study sites in Georgia and Alabama. This analysis showed that 38 of the 49 (78%) estimates were reasonable according to Guthery and Lusk's (2004) standards, as opposed to only 10 of 58 (17%) in their review of published studies. Collectively, these 3 studies (Palmer and Wellendorf 2007, Terhune et al. 2007, Sisson et al. 2009), produced annual survival rates within the range of what is expected for the region based on the theory of operational constancy in bobwhite demographics (Guthery 1997). In an effort to address this subject more thoroughly, and as part of a larger on-going study of bobwhite population ecology and management in South Georgia, we examined additional data from our studies for evidence of "radio-handicapping" and discuss anecdotal information in support of this data. We examined the following: 1) variation in survival distributions among treatment (newly radio-tagged) and control (previously radio-tagged) groups; 2) proportion of trapped animals censored during the traditional 7-day censor period; and 3) harvest of banded versus radio-tagged birds. In addition, we present anecdotal information from our studies which provide further support regarding radio-handicapping of bobwhites.

Study Area

Harvest and radio-telemetry generated data for these analyses were collected on three study sites of the Albany Quail Project during 1992-2005. These were privately-owned quail plantations that were contiguous and within a matrix of similar properties each with a similar history and management program. Together, these three encompassed 15,200 ha in Dougherty and Baker Counties near Albany, Georgia. All three were typical of properties in the region as they were characterized by mature old-field pine forests (80%) with a low basal area (3-9 m^2/ha) and scattered 1-4 ha fallow fields (20%). Management techniques included maintaining an open canopy through timber thinning and mid-story hardwood removal, frequent prescribed burning, seasonal disking of fields, drum-chopping and mowing, supplemental feeding, and mammalian predator control. As a result of this management style, quail densities in recent years averaged approximately 5 birds/ha. More detailed descriptions of these sites and their management programs can be found in previous works (Simpson 1976, Yates et al. 1995, Burger et al. 1998, Sisson et al. 2000*a*,*b*, 2002*b*, Hughes et al. 2005, Terhune et al. 2006, 2007).

Methods

All of the studies on these research sites were part of an on-going study by the Albany Quail Project with research protocols similar for all sites and the data pooled by year. Wild bobwhites were trapped on active study sites twice each year (Mar-Apr & Oct-Nov) during 1992-2005 using standard, baited funnel traps (Stoddard 1931). Each bird was classified by age and sex, weighed, leg banded, and a sub-sample was radio-tagged and released at the capture site. Only birds weighing ≥132 g were outfitted with pendant style transmitters (6 g; <5% of body weight) equipped with an activity switch (Holohil Systems, Ltd., Ontario, Canada). Trapping, handling, and marking procedures were consistent with the guidelines in the American Ornithologists' Union Report of Committee on the Use of Wild Birds in Research (American Ornithologists' Union 1988) and the protocol was approved by the Auburn University Institutional Animal Care and Use Committee, IACUC.

We monitored all birds at least 2 times weekly using the homing method (White and Garrott 1990) and recorded all locations on aerial photographs. The Kaplan-Meier staggered-entry method was used to produce all survival estimates (Kaplan and Meier 1958, Pollock et al. 1989). Seasonal survival estimates were based on a biological year beginning 1 October and ending 30 September the following year. This annual period was divided into 2 seasonal intervals for analysis as described by Burger et al. (1998). The fall-winter interval (1 Oct - 31 Mar, 182 days) began with termination of nesting and formation of coveys. The spring-summer interval (1 Apr - 30 Sep, 183 days) began with covey breakup and initiation of mating. An effort was made in every case to insure that birds were evenly distributed across the landscape during trapping and radio-tagging to reduce among covey bias (Pollock et al. 1989, White and Garrott 1990). We used the traditional 7-day conditioning period where birds that died within 7 days of radio-tagging were excluded (i.e. censored) from the analysis (Kurzejeski et al. 1987, Pollock et al. 1989).

The potential effects of capture, handling, and radio-tagging of bobwhites was evaluated by comparing Kaplan-Meier survival distributions for treatment (newly radio-tagged) versus control (previously radio-tagged) bobwhites during 8 post-trapping sessions (i.e., alternating periods of fall-winter and spring-summer) from 2000-2004. A 30-day post-trapping survival curve with no censor period was used for newly tagged birds and was compared to the same time period for birds still being monitored from a previous trapping session. Only previously tagged birds that had been radio-tagged and monitored for a minimum of 90 days were included as controls in this analysis.

The influence of the traditional 7-day censor period on end-point estimates was evaluated by comparing Kaplan-Meier survival distributions for newly radio-tagged individuals for 10 trapping sessions during 2000-2005. The comparison was made between seasonal survival estimates with and without birds that would have been censored during this 7-day period. In addition, we evaluated the magnitude of the effect censoring had on survival estimates if they remained in the analysis, and compared the proportion of birds censored in our analysis to those in previous studies.

Harvest rate was used as a measurable indicator of the effects of transmitters on cause-specific mortality. Following the methods of Parry et al. (1997) and Cox et al. (2004) we present first year (direct) recovery rates, an index to harvest rates, for birds banded or radio-tagged during the fall trapping session immediately prior to the 13 hunting seasons from 1992-2004. Reporting rates for harvest were virtually 100% because these studies were conducted on private property where hunting and harvest were completely controlled, and records of all harvested birds were put into a large research data base (Terhune et al. 2007). Thus, we calculated the simple binomial probability of recovery with no correction for un-retrieved loss.

Results and Discussion

We used a total of 1,194 radio-tagged bobwhites for our survival analysis to compare previously tagged (n = 293) versus newly tagged (n = 901) individuals. No significant difference ($P < 0.05$) was detected in Kaplan-Meier derived survival estimates for the 30-day post trapping period for all but 1 of 8 sessions during 2000 - 2004 (Table 1). In this case (spring 2003), the newly tagged birds (0.971, SE = 0.015) actually had higher survival than their previously tagged counterparts (0.878, SE = 0.042) (Table 1). This supports the findings of Palmer and Wellendorf (2007) and Terhune et al. (2007) whom also demonstrated no difference in survival between radio-tagged and banded birds. In addition, our analysis did not indicate a negative effect from trapping and handling itself since the control group was from a previous trapping session, was not caught during the current trapping period, and therefore was not susceptible to the potential effects of trapping and handling. Combined, these comparisons provide strong evidence that there were no negative impacts on survival from radio-transmitters for these study areas.

The effect of censoring on Kaplan-Meier survival estimates was negligible. These comparisons were made for 10 seasonal survival estimates from 2000-2005, and no differences ($P < 0.05$) were detected between survival curves with or without this group of birds. Of the 1350 bobwhites newly radio-tagged and added to the sample during this time period, only 18 (1.3%) were censored during the traditional 7-day period. Including these birds in the Kaplan-Meier analysis affected seasonal survival estimates by only an average of 0.016 (SE = 0.005, range 0.00 - 0.04). This is in stark contrast to estimates reported in other studies such as Cox et al. (2004) in

Table 1: Thirty-day Kaplan-Meier survival estimates for previously and newly radio-tagged northern bobwhite quail following 8 trapping sessions on Albany Quail Project study areas in South Georgia during 2000 - 2004.

	Previously radio-tagged				Newly radio-tagged			
Season	n	Surv	SE	95% CI	n	Surv	SE	95% CI
Fall 2000	18	1.000	0.000	1.000-1.000	59	0.983	0.017	0.950-1.000
Spring 2001	29	0.897	0.056	0.788-1.000	132	0.985	0.012	0.964-1.000
Fall 2001	25	1.000	0.000	1.000-1.000	120	0.992	0.008	0.975-1.000
Spring 2002	62	0.949	0.023	0.894-1.000	128	0.948	0.019	0.910-0.985
Fall 2002	24	1.000	0.000	1.000-1.000	116	0.905	0.027	0.853-0.957
Spring 2003	58	0.878	0.042	0.796-0.961	118	0.970[a]	0.015	0.940-1.000
Fall 2003	21	0.899	0.067	0.768-1.000	119	0.933	0.023	0.889-0.977
Spring 2004	56	0.924	0.036	0.853-0.995	109	0.945	0.023	0.903-0.987

[a] Survival significantly greater ($P < 0.05$) than previously radio-tagged.

which 24.4% of birds radio-tagged did not survive a 14-day conditioning period. Furthermore, Osborne et al. (1997) reported 54% of their radio-tagged sample had trouble with the harness. Our estimates are more consistent with Burger et al. (1995) who reported only 19 of 1,001 birds (1.9%) having trouble with the harness, and Burger et al. (1998) who reported censoring only 16 of 831 (1.9%) from a radio-tagged sample in Georgia during a 7-day censor period. Guthery and Lusk (2004) suggest that the routine application of a censor period was prima facie evidence of at least transient debilitation from radio-tags. We suggest the routine application of the 7-day censor period has been largely due to the recommendation of the original authors publishing the survival analysis technique used (Pollock et al. 1989), and the subsequent necessity of doing so to publish survival information when using this technique. Our analysis, combined with the discontinued use by the AQP and many other researchers studying bobwhite demographics and population ecology in the Southeast (L.W. Burger, W.E. Palmer, J. P. Carroll, personal communication) does not support the notion of an accepted censor period being evidence of ”radio-handicapping”.

We examined direct recovery rates for a sample of 3,932 banded and 2,086 radio-tagged birds during 13 hunting seasons from 1992-2004. Annual recovery rate by harvest averaged 6.68% (range: 3.4 - 11.7) for banded birds and 6.65% (range: 3.4 - 11.1) for radio-tagged birds and was not different ($P <$ 0.05) between groups during any of the 13 hunting seasons (Table 2). While these harvest rates are admittedly conservative, this analysis does not support the conclusions of previous studies that radio-tags render bobwhites more or less vulnerable to harvest than banded birds. Guthery and Lusk (2004) called into question such inferences obtained from radio-telemetry as the nature and magnitude of cause specific mortality, arguing that if radio-tags effected survival information then it made sense they were affecting other estimates as well. Due to the subjectivity and potential observer error associated with ascribing specific causes of mortalities, this topic has proven difficult to independently verify. However, harvest supplants the inherent observer subjectivity and thus provides a relatively reasonable check. Empirical studies have recently shown no difference

Table 2: First year (direct) harvest recovery rates (K) of banded or radio-tagged northern bobwhites on Albany, GA area Plantations, Baker and Dougherty Counties, Georgia from 1992-93 to 2004-05.

	Banded			Radio-tagged		
Year	n	K	SE(K)	n	K	SE(K)
1992-93	200	0.065	0.017	112	0.045	0.020
1993-94	422	0.047	0.010	282	0.067	0.015
1994-95	115	0.070	0.024	227	0.066	0.016
1995-96	98	0.112	0.032	126	0.103	0.027
1996-97	93	0.075	0.027	179	0.095	0.022
1997-98	111	0.117	0.031	190	0.111	0.023
1998-99	238	0.067	0.016	149	0.067	0.020
1999-00	652	0.041	0.008	125	0.072	0.023
2000-01	434	0.060	0.011	117	0.034	0.017
2001-02	494	0.059	0.011	189	0.042	0.015
2002-03	602	0.038	0.008	110	0.055	0.022
2003-04	234	0.034	0.012	147	0.048	0.018
2004-05	239	0.084	0.018	133	0.060	0.021
TOTAL	**3,932**	**0.067**		**2,086**	**0.067**	

in harvest rates between banded and radio-tagged birds (Palmer and Wellendorf 2007, Terhune et al. 2007) whereas other studies have produced mixed results ranging from marginally higher (Corteville et al. 2000, Cox et al. 2004) to significantly lower (Parry et al. 1997) harvest of radio-tagged compared to banded birds. This was a concern to us at the inception of our project in 1992; therefore we have tracked radio-tagged and banded birds during harvest time periods for the duration of our research program. Our results re-enforce the conclusion that radio-tagging quail does not affect their vulnerability to harvest on our study sites.

Guthery and Lusk (2004) used anecdotal evidence to explain abnormal behavior of radio-tagged bobwhites. Such accounts included radio-tagged birds less likely to flush than the non-tagged members of a covey, as well as the observation of radio-tagged birds dieing in fires while non-tagged birds escaped. These observations can be countered with innumerable observations from monitoring over 8,000 radio-tagged birds over the last 14 years on our study sites in Georgia. Our combined experiences during the course of these studies have never led us to these same conclusions. Parry et al. (1997) documented radio-tagged birds being less vulnerable to harvest on their study site in Oklahoma, and proposed this was due to their habituation to humans and reluctance to fly when encountered by hunters. Our field staff has never made such observations, nor do our studies of encounters between hunters and radio-tagged coveys support these observations (Sisson 1996, Sisson et al. 2000*c*, Sisson 2005). During these studies, our field staff monitored over 1,100 encounters with radio-tagged coveys over an 8-year period without making any observations that radio-tagged birds behaved abnormally while being hunted.

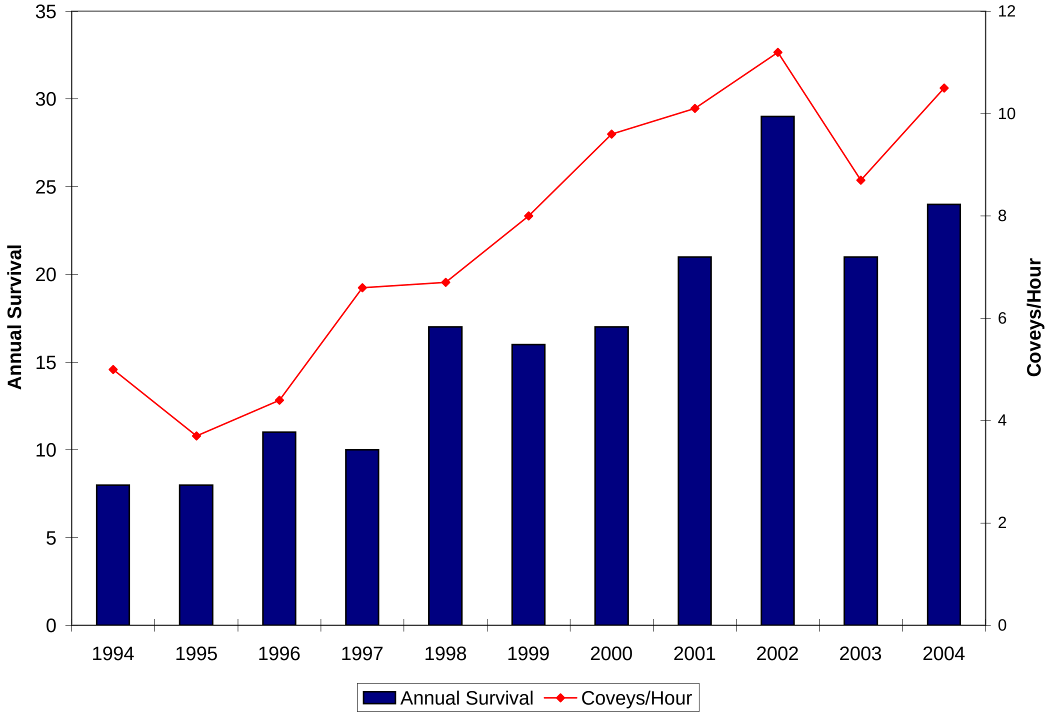

Figure 1: Kaplan-Meier annual survival estimates and coveys seen per hour hunted for one of the long-term study sites of the Albany Quail Project in Dougherty and Baker, Counties Georgia during 1993 - 2004.

Perhaps the best testament to the reliability of telemetry-generated data is whether it accurately reflects population performance of the population at large under study. Guthery and Lusk (2004) pointed out the paucity of investigations on the demographic consequences of the reported survival rates based on telemetry. Results from our study indicate that telemetry-based estimates of population parameters represent those of the population under study. This can best be illustrated by examining a case study: consider the population dynamics of one of our long-term study sites during the 11 years it was monitored year round with radio-tagged birds. We used coveys observed per hour as an index to population density (Palmer et al. 2002). Figure 1 illustrates how this population was closely associated with annual survival during the preceding year. The first 3 years of this study (1993-94 through 1995-96) were used in the meta-analysis by Guthery and Lusk (2004) as evidence of radio-handicapping due to annual survival estimates averaging only 9% during this time period (Sisson et al. 2000*a*). Annual survival rates have clearly recovered since then with a corresponding response in population density and hunting success (Figure 1). Further evaluation highlights the problem of using individual years or a short series of years (i.e., 2-4 years) in these type analyses. Four of the 11 annual survival estimates in Figure 1 are below the threshold of 0.125 proposed by Guthery and Lusk (2004) for the maintenance of population stability, while the long-term average is well above it. Very different conclusions could be drawn from these data depending on which years the population was under study. In reality, during the initial years of the study the population was declining (Sisson et al. 2000*b*,*a*), during the middle years it was responding to habitat improvements (Sisson et al. 2002*a*) and currently is relatively high and stable (Stribling and Sisson 2009).

We are uncertain at this point why "radio-handicapping" appears to be problematic for some areas and not others, but we suspect it may be related to a combination of regional differences in habitat conditions, climate, harvest pressure, and trapping and handling techniques. We agree with Guthery's interpretation of Bro et al. (1999) that the influence of radio-tagging on a species might depend on the environmental context of the population. The populations we have studied were intensively managed properties, in a region of mild climate, and subjected to a conservative harvested. In addition, our analyses were based on a large sample of birds over many years, leading to a more powerful inference about the effects of radio-telemetry on survival. We believe the direct and circumstantial evidence is compelling that we conclude that no radio-handicapping existed on our study sites and advise other researchers to follow similar approaches before drawing inferences about population parameters from radio-telemetry data.

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