A Bioeconomic analysis of wildlife management in a Natural Park:
San Rossore Estate, Tuscany, Italy

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Abstract

The management of wildlife species as pests involves making choices that determine how much pests control will cost, and what kind of benefits it will deliver. In order to make these choices defensible, the effect courses of action have on how the costs and benefits of pests control accrue should ideally be understood. This study proposes a novel approach to estimate the choice of a wildlife management of an ungulate species in a conservation site (Migliarino-San Rossore-Massaciuccoli Regional Park, Tuscany), combining biological and economical trends. In fact the management of wildlife resources provides contrasting benefits and costs, which ecological or economic approaches alone cannot analyze in their complexity and, at the same time, can only offer a limited insight.

The main problem is that, both in protected areas than in country lands (where there are regulated hunting areas), some vertebrate species are considered as pests. In these cases pests are considered as species able to create different kinds of damage to the environment in which they live.

The purpose of this work is to adopt an interdisciplinary integration of research expertise from natural sciences, economics and social sciences to manage a fallow deer population in an ex-hunting Estate in Italy, now part of a Regional Park. The aim of this work is to develop a model to achieve a balance constrained by biological and economical variables. Ecological-biological problems regarding environment and wildlife management are usually solved separately by economic tasks. Because bioeconomic control problems are still new objectives of the wildlife management in Italy, this research aims to give an overview of the classical bioeconomic models to introduce a new technique in decisions regarding wildlife species management and eventually harvesting control programs.

Bioeconomic models are central to this approach as they combine biological data about population dynamics, sex and age class segregation, habitat use by the biological population, with economical data, deriving by costs for fences to reduce environmental damages and car accidents, costs for harvesting, revenues by venison and trophy, and etc. The primary objective of this work is to produce a bio-economic framework with sufficient structural complexity to analyze the management of this fallow deer population at our local level. This objective could be achieved developing a deterministic biological model that later would be implemented on a bioeconomic one. First, we develop a model in which wildlife managers in a Park seek to balance the revenues by the culling with the costs of the management, as the Italian law restrictions require. In a second step we will try to develop a simulating model imagining our Protected Area in Italy as a Sporting Estate in which the landowner desires to maximize his profit by venison and trophy value in an ecological equilibrium. Later we will use the simulating software Vensim to manipulate our fallow deer population in all the ways and conditions we will, combining the population growth, the size of the cull, and the desired profit. Finally, three different approaches to bioeconomic wildlife management plans are analized to show new possible horizons of the wildlife management activities in Italy: an hunter's utility maximization problem; a wildlife management maximising the social welfare; a wildlife management maximising meat and trophy value in ecological equilibrium.
This work provides techniques to people managing conservation and exploitation of environmental resources to realize the optimal balance between all the variables acting (ecological, economic, social,..).

*Keywords:* Bio-economic model, Ungulates, Fallow Deer, Natural Park, Deterministic model, Optimal control management, Pests.
1. Introduction and literature review

Pests can be defined as organisms that cause harm: environmental, economic, or epidemiological. Cherrett et al. (1971) defined a “pest animal” as “one which is noxious, destructive or troublesome to humans”. Woods (1974) qualified the definition to incorporate a requirement of causing economic damage. Today we complete the definition observing that there are some vertebrates causing problems in agriculture: to crops, to livestock, in and around buildings and other equipment, and spreading diseases. Problems also can occur in forestry, in conservation of plant and animal species and communities. For example, imagine a scene of many square kilometres of semi-arid scrublands where an isolated population of 1000 goats is eating the local shrubs. The scene could be interpreted in many ways. In their native environment, or one long managed for agriculture, the goats may be called indigenous or domestic animals respectively. In other parts of the world the goats may be called pests. What is the difference? It is the effect that the goats have on the scrublands rather than the goats themselves that can be annoying and may necessitate pest control. Hence, some scientific assessment of the effects of goats is needed.

In a broader context, it is the damage of vertebrate pests that justifies their economic control.

For effective pest control, it is fundamental to determine the pest status of an animal. Until this is established, a statistical or economic evaluation of damage is pointless. In a general way, we can decide to take like guide-line the following eight reasons that Judenko (1973) listed for assessing the losses caused by pests: (a) establish the economic status of specific pests, (b) estimate the damage or abundance level that justifies control, (c) calculate the justifiable expenditure of control, (d) estimate the effectiveness of control, (e) measure the effects of environmental factors on the loss of yield caused by pests, (f) give information to manufacturers and distributors of pesticides to help them to decide what actions should be taken, (g) assess the public use of funds for current research, (h) direct future research and planning. Auld and Tisdell (1986) expanded on points (c) and (h), by arguing that it is only rational to increase our knowledge about pests up to the point where the marginal benefit from research equals the marginal cost of obtaining that extra knowledge.
Another point to be considered is that damage by vertebrate pests can vary in space and time. More emphasis in vertebrate pest research is, however, on the mean level of damage. Before consider the main aim of the work, it could be interesting briefly to explain into two words the patterns of such variation. Spatial variation is the frequency distribution of damage between areas at the same time. This variation can be examined at different geographic scales, such as within fields or forests, or in a district or county, or a region or a country. The distribution can vary from negative exponential to normal, to positively skewed (Fig. 1.1)

(Fig. 1.1 Possible frequency distributions of spatial variation in damage by vertebrate pests. (a) Negative exponential, (b) normal, (c) positively skewed, (d) bimodal. In each example, the x axis gives the level or extent of damage, and the y axis gives the number of sites with each level of damage (the frequency)) (Figure adapted from Hone, 1994)

Each unimodal (one-peaked) frequency distribution is an example of a more general exponential frequency distribution (Dobson, 1983). Bimodal (two-peaked) or multimodal (many-peaked) distributions could also occur, though only few literature reports information about these conditions. It could be interesting to have a fast overview of some of these frequency distributions. Dawson (1970) reported a positively skewed frequency distribution of damage to grain crops by sparrows (Passer domesticus) in parts of New Zealand, and Buckle, Rowe and Husin (1984) cited a study of rat damage to rice fields in Malaysia that showed a log-normal frequency distribution of damage. Negative exponential frequency distributions of damage have been reported for damage by feral pigs (Sus scrofa) in south-eastern Australia (Hone, 1988a). But many others are the examples that could be cited about
damage by herbivores (Moore et al., 1999; Hester et al., 1999; Gordon et al., 2004; Zivin, 2000, and more over).

This introduction about pests is to give more general conclusions: many wildlife populations could cause damage to agricultural and forestry systems, but they are also valued as resources, either for their recreational value or for their value of existence and contribution to biological diversity. As a result, the economic and regulatory environment in which the species exists, whether they are considered as “pests” or as “resources”, often determines the “real nature” of given species.

In this study different bioeconomic models are developed: the aim of the main case study of the present work is to investigate biological and economic mechanisms driving an “optimal control strategy” to manage a fallow deer (Dama dama) population in a Natural Park in a sub-Mediterranean habitat, where large predators are absent. This wild population well represents the double face of the problem, as it is at the same time a pest and a natural resource. In the first case it causes damage to forestry not only because natural predators are absent, but also because two rivers (the Arno river at North and the Serchio river at South), the Tyrrhenian Sea and a big roadway bound the natural protected area within a strongly urban land, preventing any possible emigration out the Reserve. At the same time this species is the most representative mammal of the Park, for its biological and recreational status. But which is the border to consider animals as pests or not? Surely it is important to consider which kind of land we want to manage. Fallow deer population (as other species in the Park) have, in fact, a use value and a non-use value associated with it. What does it mean? Briefly the use value is, for example, generated by benefits arising from venison and trophy price, and costs due to damage to agricultural crops and forestry, and its control. The non-use value derives by a recreational value that each different people can give to wildlife, or by the simply existence of these species like part of a biological diversity.

Some more general considerations could be arisen about the value associated to natural resources. A range of values could be attributed to natural and environmental resources, such as landscapes in general, woodlands in particular, and also wildlife within landscape, deers, birds, wolves in woodland. Some of these values derive from the fact that natural resources produce or provide marketed goods or services to society, either directly or indirectly. Goods and services such as softwood timber from plantation forests, agricultural crops, and recreational amenities for which
Entry fees are levied, are traded directly in financial markets, and the value which they provide to society can be determined by observing how quantities purchased vary as the price of the goods concerned change (Kolstad and Braden, 1991). Environmental and natural resources also provide other goods and services which are used by society, but for which no trading market currently exists (Krutilla, 1967). Such goods and services may arise indirectly from the existence of woodland, deers or other wildlife species within the landscape.

The value generated directly or indirectly by the goods and services mentioned above are termed *use values*, since actual contact is required with the natural resource that provides good or service in order for “value” to be extracted. Observable financial transactions are not associated with the provision of all direct use values that are produced by natural and environmental resources, and financial markets do not usually exist for supply of indirect goods and services from these resources. The values attached to these goods and services in such circumstances are termed *non-market values*. It can be difficult to establish the value that society derives from the provision of products of these types, but non-market and indirect use values can sometimes be established by observing expenditure on related market goods (Maler, 1974; Kolstad and Braden, 1991). For example, the travel expenditure incurred by a cross section of visitors to a recreation site could be used to derive an estimate of the minimum value of the site to society at large (Clawson, 1959). Appropriate examination of the impact of scenic views or proximity to forest landscapes on house prices could also allow a minimum value for these environmental resources to be determined (Brookshire et al., 1982). Environmental and natural resources can also generate value to members of society who have never encountered the resources directly, and may never intend to encounter them at some point in the future (Krutilla, 1967). Values of these types are called *non-use values*. Non-use values may accrue to much larger sections of society than solely the residents of, or visitors to, the resource concerned, and the non-use value of natural resource to society as a whole might therefore be substantial, even though the non-use value of the resource for each individual is only small (think about a Natural Park and all the benefits deriving by its environment).

To understand how costs and benefits vary among different pests management strategies, the biological and management components of a pest/resource system
must be linked so that its economic inputs and outputs can be estimated and compared.

Clark (1976; 1990) coined the term “bioeconomics” to describe the economic analysis of biological systems, and described a conceptual and mathematical framework that links the dynamics of biological populations with the economic imperatives that drive their management. Clark (1990) placed emphasis on the use of analytical models to predict the effect of management on system behaviour, and this one has recently been extended to include dynamic programming and simulation/optimization approaches that are useful when system behaviour is stochastic or uncertain (Choquenot and Hone, 2002; Williams, 1989; Hilborn and Mangel, 1997; Alexander and Shields, 2003). However, regardless of how they are actually implemented, the principles of bioeconomics provide a potentially powerful approach to the analysis of pest management systems. Given this potential, it is surprising how few examples of bioeconomic analysis have been reported for wildlife pest management (Hone, 1994), particularly for wildlife management in Italy.

Concerning the modelling of endangered species, it was developed principally out of the literature of fisheries economics. Anyway the basic model was still developed by Clark (1973): he based his model of species extinction on Gordon’s (1954) seminal fisheries model in order to examine the conditions under which the elimination of a species may appear to be the most attractive policy to a resource owner. This model, in fact, forms the foundation of much of the species extinction research that has followed. More recently, Swanson (1994) examines ways of making this model more readily applicable to terrestrial species by generalizing the analysis to consider terrestrial resource allocations. He provides a theoretical framework for the economics of extinction that consider the elimination of species as a result of human choice. Swanson argues that mankind has a “portfolio” of productive assets and that substitutions are made between these assets (less productive assets being replaced by more productive assets) through a process of investment and disinvestment. The marginal productivity of a biological asset generally declines as population levels approach carrying capacity; it means that there is typically some identifiable optimal level of species stock in productivity terms. Whenever returns to the asset are less than the market rate of return, disinvestment will occur. In the case of endangered species, this disinvestment often
takes the form of the reallocation of the primary resources required for species survival. Swanson’s model implies that the stock of a particular resource will move to the level that equates its rate of return to that of other competing assets in our portfolio. Extinction is seen as a complete disinvestment of a wildlife resource, which occurs because it is perceived as not being worthy of investment (Swanson and Barbier, 1992). These and similar models have generated important insights into the behaviour exhibited by resource managers, property owners and harvesters of open-access resources, but they are limited in the characteristics of species to which they apply. For example, both the Clark and Swanson models consider only the consumptive value of the species in question. Many other models have been developed that add tourism value and even existence value, but they are still driven by harvest, both as principle means of value generation and as the variable through which a population is controlled by resource managers (Bulte and van Kooten, 1999; Alexander, 2000). While there is a great number of species for which these approaches do apply, there are many others for which they do not. The question arises as to the applicability of such bioeconomic models to species for which there is no harvest value. In such a case, a wildlife manager should have to consider 1) the nature and magnitude of values generated by the species, and 2) the nature of the control humans exercise over that species.

Anyway this last case is not actually part of our study, which is more close to Gordon’s model and to the theory of the existence of a “bionomic” equilibrium. The analyses described in this work are based on models qualified by ecological and economic variables; in other words the framework of the model is developed to use interactions between wildlife population control, fallow deer density and distribution, to contrast the economic inputs to ungulate management (costs of controlling deer population), with its economic outputs (the recreational value for people visiting the place, arising by the simple existence value that this species has in forestry ecosystem, and the value of benefit that accrue from control through the venison sale) (Fig. 1.2).
(Fig.1.2. The general structure of the bioeconomic model developed in this work. The bioeconomic model, which can have varying levels of complexity, is used to link economic inputs to the management system (in this case costs of harvesting the ungulate population, and by forestry damage) and consequent economic outputs (in this case the benefits arising by the control of the wildlife populations). These economic inputs and outputs are analyzed to identify pest control strategies that produce some optimal outcomes)

The key assumption behind all sustainable harvesting models is that populations of organisms grow and replace themselves; that’s why they are called renewable resources. Otherwise, it would be impossible to harvest them sustainable, and they would be treated theoretically in a similar way to reserves of fossil fuels. Some species, such as Antarctic blue-whales (Balenoptera musculus), for examples, may grow so slowly that, to all intents and purposes, harvesters should treat them as non-renewable resources.

Another assumption of renewable resource harvesting is that populations of organisms do not continue to grow indefinitely; they reach an equilibrium population size, which occurs when the number of individuals matches the resources available to populations. The population size remains stable at this size. The equilibrium size of the population is its carrying capacity ($K$).

The management of renewable resources has generally been based on the concept of maximum sustainable yield (MSY). This could be considered, probably,
the simplest yield management objective that accounts for the fact that a biological resource stock cannot be exploited too heavily without an ultimate loss of productivity. The concept of MSY is itself based on a model of biological growth (Fig. 1.3), which assumes that at any given population below a certain level of $K$, a surplus production exists that can be harvested in perpetuity without altering the stock level.

![Surplus production equals sustainable yield](image)

(Fig. 1.3 The concept of Maximum Sustainable Yield)
(Figure adapted from Clark, 1990)

If the surplus is not harvested, on the other hand, corresponding increase occurs in the stock level, which ultimately approaches the environmental carrying capacity $K$, where surplus production is reduced to zero. Moreover, since surplus production equals sustainable yield at each level, it follows that MSY is achieved at the population level where surplus production is greatest (i.e. at the level where the growth rate is maximized). For most populations to which this model applies, the MSY level is found to lie between 40% and 60% of the environmental carrying capacity (Getz and Haight, 1989; Clark, 1990). But we have to pay attention about situations in which commercial exploitation may lead to serious overexploitation of biological resources in the sense that the population becomes reduced to a level far below MSY. There are convincing reasons why such an outcome, which may actually be “optimal” from the point of view of resource exploiters, may be far the optimal from the biological and social viewpoint. Recognition of the inadequacy of the MSY concept has resulted in a trend to replace it with some concept of optimum sustainable yield (OSY). In this work we try to use the concept of optimal resource
management basing it on the standard cost-benefit criterion of maximizing present values of net economic revenues.

The economic theory of open-access resources was developed by H.S. Gordon in 1954. This theory predicts an ultimate “bionomic equilibrium” between the resource and the exploiting firm (or industry), which occurs at a stock level where the revenue exactly equals the cost (opportunity cost) of exploitation. This analysis is of particular importance for our case study: as we observed previously, the model developed for wild population harvesting in a Natural Park in Italy have to take in mind the Italian country policy for Protected Natural Areas under which wildlife populations are “…exclusive patrimony of the State” (L. 394/92), and a Natural Park cannot be analysed as a farm or a business firm whose main aim is to maximize total profit. At the same time, the decreasing of funds by the Italian Government to Parks and other Natural Protected Areas could change the horizon of wildlife management plans and, Parks as first, could play a central role in this new “environmental” business using all the environmental sustainable activities allowed in Protected Areas to develop and manage the same one and to give more opportunities of way of life to new generations.

Aiming to explain different approaches to wildlife management plans of action under biological constraints and in a biological-economic perspective, the present work provides different bioeconomic models, showing theoretical and empirical conditions for optimal managements.

The following work is organized in different steps. Firstly, a biological model using the Leslie Matrix method, and later two bioeconomic models are developed in a Protected Area in Tuscany (North-Central Italy), where an overabundant fallow deer population is considered both as pest as resource. Moreover, different bioeconomic wildlife management approaches, used in many countries, will be analyzed to offer new tools to change the Italian wildlife management horizon. Finally, a brief overview of the simulation software Vensim will be analyze.

2. Dynamics of exploited populations

We will describe now a simple model of population dynamics of a single species. The model will be then modified to include harvesting either a fixed number of individuals or a fixed proportion of the population. It is important to remember
that different harvesting strategies and harvesting levels produce equilibrium population sizes that differ in stability. The usefulness of the concept of maximum sustainable yield (MSY) will be discussed, and it will be demonstrated that it flows both from biological and economic perspectives.

The first step is to develop the logistic growth model (it seems that P.F. Verhulst was the first to use this model, in 1838, as model of human population growth; later the model was modified by Pearl, 1930; by Gause, 1935; etc.) for a renewable biological resource. We start supposing that in a certain population both the birth rate $b$ and the mortality rate $m$ are proportional to the population size.

So the net proportional growth rate will be the simple equation $r = b - m$.

By the previous equation we obtain the differential equation

$$\frac{dx}{dt} = rx$$

that is a continuous time model of population growth (consider that $\frac{dx}{dt}$ could also be written as $\dot{x}$). As it is clear, the solutions to this equation $x(t) = x(0) e^{rt}$ grow exponentially to infinity if $r > 0$ and decrease exponentially if $r < 0$. In an ideal world, where no spatial or temporal constraints inhibit growth, many biological populations are observed to grow at an approximately exponential rate initially. Anyway such a process cannot proceed indefinitely. As the population level $x$ increases, some environmental limitation must force the proportional growth rate to decline. To understand this effect the Eq. (1.1) has to be modified in the form

$$\frac{dx}{dt} = r(x) \cdot x$$

where $r(x)$ is some decreasing function of $x$.

The proportional growth rate becomes

$$r(x) = \frac{F(x)}{x}$$

which now depends on the population level $x$.

As $r(x) = \frac{F(x)}{x}$ is a decreasing function of $x$, this simple model is said to describe a process of feedback, or in other ways of compensation, which controls the growth of the population as its level increases.

The most used equation for population dynamics is surely obtained when
\[ r(x) = r \left(1 - \frac{x}{K}\right) \]
so that the Eq. (1.2) becomes

\[ \frac{dx}{dt} = rx \left(1 - \frac{x}{K}\right) = F(x) \]  

(1.4)

The last equation is the so-called logistic equation. The constant \( r \), assumed to be positive, is the intrinsic growth rate, since the proportional growth rate for small \( x \) approximately equals \( r \). The positive constant \( K \) is usually referred to as the environmental carrying capacity, or saturation level (population or environmental saturation level).

The Eq. (1.4) possesses two equilibrium solutions, at \( x = 0 \) and at \( x = K \). We should have consider that

\[
0 < x < K \text{ implies that } \frac{dx}{dt} > 0, \quad \text{whereas } x > K \text{ implies that } \frac{dx}{dt} < 0.
\]

It is easy to understand that \( K \) is a stable equilibrium, or, to be more precise, it is globally asymptotically stable for positive \( x \) in the sense that

\[
\lim_{t \to +\infty} x(t) = K, \quad \text{provided that } x(0) > 0.
\]

(Fig. 2.1 The logistic equation: (a) the logistic growth function \( F(x) = rx(1 - x/K) \); (b) typical solution curves) (Figure adapted from Clark, 1990)

This condition is illustrated in Fig. 2.1, in which part (a) shows the growth function \( F(x) = rx(1 - x/K) \) and the arrows indicate the direction of change of \( x(t) \) with increasing of \( t \). Figure 2.2.b illustrates two typical solution curves \( x(t) \).
approaching the equilibrium point $K$ from above and below. The lower curve, with its characteristic *ogive shape*, is usually referred to as a *logistic growth curve*.

The logistic differential equation Eq. 1.4 is easily solved by separation of variables.

In fact we can write the Eq. (1.4) in the form

$$\frac{dx}{x(K-x)} = \frac{r}{K} dt$$

or

$$\left(\frac{1}{x} + \frac{1}{K-x}\right) dx = r \cdot dt$$

so that, by integration

$$\ln \frac{x}{K-x} = rn + \ln \frac{x_0}{K-x_0}, \text{ where } x_0 = x(0).$$

The solution may be rewritten in the form

$$x(t) = \frac{K}{1 + ce^{-rt}}, \text{ where } c = \frac{K-x_0}{x_0}.$$

(1.6)

We can now suppose that the population described by the logistic equation (Eq. 1.4) is subject to harvesting at a rate $h(t)$. Then the logistic equation becomes

$$\frac{dx}{dt} = F(x) - h(t), \quad (1.7)$$

where

$$F(x) = rx(1 - x/K).$$

Many observations could be made about the dynamic behaviour of the population.

An important special case occurs when $h(t) = h \equiv \text{constant}$:

$$\frac{dx}{dt} = F(x) - h. \quad (1.8)$$

In case $h < \max F(x) = \frac{1}{4} rK$, the Eq. (1.8) possesses two equilibria, at $x_1$ and at $x_2$ (see Fig. 2.2a).
We can observe that \( \frac{dx}{dt} > 0 \) when \( x \) lies between \( x_1 \) and \( x_2 \), while \( \frac{dx}{dt} < 0 \) elsewhere. It follows that \( x_2 \) is a stable equilibrium point and that \( x_1 \) is an unstable equilibrium point. If the initial population is at \( x = K \), for example, then \( x(t) \) will converge asymptotically under constant harvesting to the equilibrium \( x_2 \). But if the initial population is less than \( x_1 \), the \( x(t) \) will approach 0. In the latter case the approach is not asymptotic (since 0 is not an actual equilibrium point of Eq. 1.8, although 0 is surely a biological equilibrium); instead \( x(t) \) is reduced to 0 in a finite time.

If \( h > \max F(x) \), as shown in Fig. 2.2b, the population approaches 0 for any initial level of \( x(0) \). Finally, in the special case when \( h = \max F(x) \), there is a single equilibrium point at \( x_1 = K/2 \), which is “semistable” in the sense that \( x(t) \rightarrow x_1 \) if \( x(0) > x_1 \) but \( x(t) \rightarrow 0 \) if \( x(0) < x_1 \).

The model given by the Eq. 1.8, although many limitations exist, provides a number of significant predictions concerning the harvesting of renewable resources.

First, there exists a maximum sustainable yield (MSY) representing by

\[
h_{\text{MSY}} = \max F(x),
\]

and characterized by the property that any larger harvest rate will lead to the depletion of the biological population (eventually to zero).

Second, the population level \( x = x_{\text{MSY}} \) at which the productivity of the resource is maximised is not the natural equilibrium level \( K \); in this model it is only half that level. It means that there is no sustainable yield at the population level \( x = K \).
Third, if \( x \) has been reduced to a level below the MSY population \( K/2 \) by some process of overexploitation, then recovery population to the MSY level requires an initial harvest rate below MSY. The most rapid recovery is achieved by \( h = 0 \).

These simple phenomena have to be considered a biological point of view about which appear to be typical of most practical resource harvesting situations, conditions that are often misunderstood by public policy and inexperienced managers. Most biologists consider a resource to be overexploited only when the population has been reduced to a level below the maximum sustainable yield \( x_{\text{MSY}} \). At the same time an obvious economic consideration could arise: the costs of catching a wildlife population tends to rise as the population is reduced.

In addition to data pertaining to the catch, fishery statistics (Schaefer, 1954; Clark, 1990), which normally include information under the heading fishing effort, measured in units appropriate for the fishery in question, have also to be considered in the model. In this case the unit of measurement could be simply the total number of vessel-days per unit time; in other cases more detailed information about the number of lines, traps, as well as the number of people employed and their wage, or the hours of work necessary for harvesting, could be available and used in the model.

The ratio of catch divided by effort is almost always taken as at least a rough indication of the current stock level of the fish (or other wildlife) population. Here we will use the phrase catch-per-unit-effort (CPUE) hypothesis to describe the proportionality between the catch-per-unit-effort and the stock level, or that

\[
h = qEx \quad \text{or} \quad Y = h = h(x, E)
\]

where \( E \) denotes effort and \( q \) is constant and is called the catchability coefficient.

By definition, a production function defines the rate of output of a certain productive process in terms of inputs to the process (Chiang and Wainwright, 2005).

The Eq. (1.9) is a special case of the Cobb-Douglas form \( Y_t = q X_t^\alpha E_t^\beta \), where the coefficients are \( \alpha = \beta = 1 \), which we will analyze in more details later.

Substituting Eq. (1.9) into our basic harvesting model, Eq. (1.7), we obtain a new equation

\[
\frac{dx}{dt} = F(x) - qEx = rx \left( 1 - \frac{x}{K} \right) - qEx. \tag{1.10}
\]
In the simplest way we can consider the solutions of the Eq. (1.10) under the assumption that $E$ is constant. To obtain the equilibria of the Eq. (1.10) we have to set $dx/dt = 0$. For any $E < rq$, we find that the equation has a unique nonzero equilibrium $x_1$, given by

$$x_1 = K \left(1 - \frac{qE}{r}\right). \quad (1.11)$$

The logistic model with constant rate of effort $E$ is shown in Fig 2.3.

(Fig 2.3 Logistic model with constant rate of effort)
(Figure adapted from Clark, 1990)

The equilibrium harvest, also called sustainable yield $Y = h$, corresponding to $E$ is given by

$$Y = qEx_1 = qKE \left(1 - \frac{qE}{r}\right), \quad (1.12)$$

provided that $E < \frac{r}{q}$. The graph of this condition is a parabola (Fig. 2.4) and represents the yield-effort curve for the model given by the Eq. (1.10). As said previously this model is commonly called the Schaefer model, as the biologist M. B. Schaefer studied it extensively and applied it to various fish populations (1954; 1957; 1967).
The smooth nature of the relationship between effort and sustainable yield in the Schaefer model should be observed. With increasing levels of effort, the sustainable yield rises to a maximum level (at $E = r/2q$, $x_1 = K/2$) and then declines equally smoothly to zero (at $E = r/q$, $x_1 = 0$). A smooth relationship between effort and sustainable yield is highly desirable in actual fisheries, since it implies that “incremental” methods can be successfully applied to management policies. Unfortunately, while many fish populations appear to exhibit a smooth yield-effort relationship, many other wildlife populations have suffered sudden collapses under the pressure of heavy harvesting rate.

3. Bioeconomic approaches to wildlife management

3.1.1 Biological and bioeconomic modelling of fallow deer in a Regional Park in Italy

The present work aims to analyze the ecology of a population of fallow deer in a reserve into a Regional Park in central Italy (the 4.653 ha study area lies on the Tyrrhenian Sea coast, West of Pisa, Italy) which, in absence of natural predators and without the possibilities to dispersal outside the Park, has reached high density causing damage to forestry and biological crops inside the Park. This condition makes necessary a managing plan to control the population, regulating both the number of animals, making it sustainable for the environment, and the costs of the Park to control the same population and the damage caused by its overabundant...
density. As the fallow deer population, which we consider, lives within a Park (the situation will be different if we have to manage a wild population in a private property, e.g. a sporting estate) the management plan is under Italian law restrictions on wildlife management plans, which no provide profit by the wildlife management and consequently business behind it. The reserve in which the current research work took place is an old hunting Estate, holiday house of Kings and Presidents in the past; it became part of Migliarino-San Rossore-Massaciuccoli Regional Park only in 2000. The Estate is bounded by the Serchio river to the North, the Arno river to the South, the Tyrrenian Sea to the West, and fenced in on the East to divide the area by a big roadway; in these circumstances, no emigrant or immigrant flows are provided.

Invasive species policy is either explicitly or implicitly underpinned by the question “When should investment in managing the invasive species stop?”. We explained in the previous section that bioeconomic modelling provides a quantitative framework for considering the benefits and costs of alternative levels of investment in invasive species management by linking the level of investment to the costs of control and value of benefits derived. Control costs are typically the product of the number of individuals that have to be removed to either eradicate the invasive species or constrain it at some specified density, and the cost of removing each individual, which will vary with population density.

Impact functions take a variety of forms, but in general are systematically related to the density of the managed population. Where impacts can be accounted in monetary terms (e.g., where an invasive species affects income), control costs and benefits can be directly compared and an optimal level of investment (usually that which maximises return on investment) can be identified. However, where impacts do not have a directly accessible monetary value (e.g., where an invasive species affects conservation values), benefits and costs cannot be directly contrasted. Under these circumstances, bioeconomic modelling can be used to identify management strategies that maximise the level of benefit, and which can be achieved for expenditure of a fixed budget (benefit maximisation), or minimise the cost of achieving a given level of benefit (cost minimisation).

Anyway, when we are developing a harvesting management plan for a certain species causing a particular impact on environment, it is important to consider the two basilar components of exploitation: the ecology of the species being exploited
and the decision-making of the people that exploits them. Both of these components need to be taken into account when modelling exploitation; only when both are included in the plan of control we can answer to questions such as: “what will the size of the exploited population be in the long term?”, “how many hunters, or effort (in general), will there be?”, “how many individuals have to be harvested every year?”. As we said in the previous section, when the biological and economic components are combined in a model of the system, it is called a bio-economic model (Milner-Gulland and Mace, 1998).

The biological component of the bio-economic model describes how the population grows, both naturally and under exploitation. The economic component describes how much effort will be put into harvesting the population. This depends both on the biological characteristics of the species and on the costs of harvesting and the price received for the products of harvesting. As the decision-making of the harvesting is usually expressed in monetary terms, the aim of the control program will be to maximise the profit. But there are situations, like the case study presented in this work, where local law do not allow to Natural Parks to maximise their profit by wildlife management. Here the main aim become to find the best population density to balance revenues by culling and costs of control, caused by animal damage to forestry and crops and, at the same time, to maintain a sustainable number of animals representing a form of social welfare benefit.

In other terms, a common goal for the management of these renewable resources, whether publicly or privately owned, is to determine the harvest rates (or rotation age for forestry) that maximise the value of the resource considering all the appropriate costs and benefits (Van Kooten and Bulte, 2000).

The biological model presented in this first approach is underlined by the “Leslie Matrix Model”, which is able to offer a prediction of population dynamics and growth through the time under a natural growth, so when no culling occurs. The aim is to show that in some places (where no natural predators are presented, where there are fences and boundaries avoiding emigration flows, and so on) a natural growth without any kind of human control (by culling) could steer to overabundant density of that species respect the carrying capacity of the area, resulting, first, in high damage to forestry and other environmental components, and in spreading diseases among the same animals. Moreover, a bounded wild population under no regular culling control programs is steered to alter the structure of its sex and age classes in
a certain timespan. Considered the charismatic aspect took on by this species in the Park considered, and that the aim of the Natural Protected Area is not only to control the damage by deer population, but also to maintain the population at densities sustainable for environmental conditions, optimal control programs could be the right way for policy maker to join ecological-social and economic point of view in a future perspective.

The annual census of the animals presented in the Estate is usually carried out in the first ten days of April, when deers are counted from high seats in pasture and woodlands, and counts are repeated at dawn and early morning for four days. Deer density within study area was established to be about 26-30 deer/100 ha.

The Park policy aims to reduce the fallow deer population at a density of about 12 deers/100 ha, and it could be probably necessary to decrease the population density until to 7 deers/100 ha (Casanova et al., 1988). That’s the reason as in the present work we will consider the different ecological-economic simulation scenarios depending on different population densities we desire to have in the Estate to achieve a sustainable balance between animals and environment.

In the present work ecological and economic data were collected from 2001 until 2006. We classified fallow deers into four classes: adult females (>12 months, so more than one year), also called as does or hinds; adult males (>48 months, so more than two years), also called as bucks; yearling males (12-24 month olds), usually called as yearlings; subadult males (24 - 48 month olds, so between two and four years), also called as prickets; all the animals (0-1 year old males and females) are called as fawns, or calves.

The following table (Tab. 3.1) shows the times during which census and removing control programs are carried out, as well as new offsprings are born.
During the six years of fieldwork, we collected data on the number of animals counted in April and noted, from October to February, the number of shot and harvested alive animals. The price of the animal is considered in terms of Kg/€, but previously the animal weight price is fixed in a public auction sale. The Estate managing plan doesn’t use different prices for males and females: it is not so surprising if we think that, from one hand, the high number of animals to remove are females and, on the other hand, the private landowner that will buy the animals to sell them again to hunters (in his private hunting Estate) prefers to buy males with big trophy which have an higher market value. The consequent agreement consists to apply the same price to all the animals as the aim is to find the higher number of buyers to remove a good number on deers.

All the costs relating harvesting actions were also collected: costs for people employed, costs to repair fenced capture sites, to repair fences for agriculture crops, costs regarding assurance after deer-vehicle collisions. Every year the Park has to sustain these costs for wildlife management.

The biological population dynamic model was carried out using the Leslie Matrix, as explained above. As fallow deer population is not an annual species, and as we consider four age-classes in which the older age-class for males and females contains animals of two (for females) or of four (for males) and more year olds, we decided to use the particular Leslie Matrix with a composite age-class for individuals, one for two-year olds for females, and one for four-years old for males, and older.

The difference in the classification of female and male age classes depends on the difficulty in recognize females older than two years, and it makes necessary for us to use two different Leslie Matrices.

Age or stage-structured population models have long been used in modelling of harvested wildlife populations, particularly fish and forestry (Levin and Goodyear, 1980; Getz and Haight, 1989). These models are linear and deterministic (Beddington and Taylor, 1973; Doubleday, 1975; Rorres and Fair, 1975), progressing later to models that included density dependence (Reed, 1980; Getz, 1980; Jensen, 1996, 2000), seasonal effects (Getz, 1980), environmental effects on vital rates (Milner-Gulland, 1994; Pascual et al., 1997; Xie et al., 1999), and spatial structure (Brooks and Lebreton, 2001). Optimization of harvest is usually carried out.
with the purpose of maximizing sustainable yield, and so equilibrium dynamics are a common focus (Scarpa, 2003). As more complicated models have been developed, it has become common in the last years to study and simulate the population dynamics under different scenarios as an alternative to formal optimization (Choquenot, et al. 2004). When we attempt to manage a natural renewable resource, it is important that the exploitation of a part of the resource/pest considered (causing damage, or spreading disease, etc..) will be a sustainable yield for the same resource (Neher, 1990). In other terms, we cannot reduce the natural resource under the minimum sustainable number of individuals under which the same population will grow to extinction. Under these circumstances, it becomes helpful to examine the two components of exploitation: the ecology of the species being exploited (Begon et al., 1996), and the decision-making of the people that exploit them. Both of these two components need to be taken into account when modelling exploitation, and after they are combined in a model system, a bio-economic model, we are able to answer to questions like: what will the size of the exploited population be in the long term?; how many individuals have to be removed to maintain population dynamics and the other ecological and economic components taking in act balanced?; which are the costs of harvesting by landowner or by Natural Park policymaker?; which are the revenues derived by the exploitation of the natural resource?.

The biological component of the bioeconomic model, describes how the population grows, both naturally and under exploitation. The economic component describes how much effort will put into harvesting the population.

The most common mathematical tool to construct a biological model (particularly if there are lacks in data, and we cannot use a life table which requires more details) is the “Leslie Matrix” (Leslie, 1945). The following matrix represents the Leslie Matrix:

\[
L = \begin{pmatrix}
F_0 & F_1 & F_2 & F_3 \\
S_0 & 0 & 0 & 0 \\
0 & S_1 & 0 & 0 \\
0 & 0 & S_2 & 0
\end{pmatrix}
\]
where $F_x$ and $S_x$ are, respectively, the fecundity and the survival rate of the $x$-year-olds, as we discussed previously. Often $F_0 = 0$, as the first age-class is a non-reproductive class in many species. The Leslie Matrix has a very specific structure. The elements on the top of the row are fecundities. The survival rates are in sub-diagonal of the matrix. All other elements of the matrix are zero. In matrix notation it becomes

$$N(t+1) = L \cdot N(t)$$

where $L$ is the Leslie Matrix. As before, we may also assume that (1) the population is closed; (2) the vital rate is constant; (3) there is no density dependence.

The Leslie matrix is a useful tool to predict the age distribution in future, simply multiplying the same matrix with the next year’s age distribution vector. These projections are made ignoring all forms of stochasticity. The fluctuations in abundance of various age classes do not result from variation in matrix elements (survival and fecundities), but from the particular distribution of individuals among age classes. Repeatedly multiplying an age distribution by a Leslie Matrix with constant elements tends to draw it into a special configuration known as the stable age distribution. Before it reaches the stable age distribution, the population may show considerable fluctuations (we are considering that these fluctuations are not caused by changes in the environment, as it could also be possible, but result from the distribution of individuals within the population).

To understand what happens to a fallow deer population in a bounded Park, without natural predators and under no culling programs, we started to use a program based on Leslie Matrix model over twenty years. We considered the series of data using as starting point the year 1986, as shown in the following table:

<table>
<thead>
<tr>
<th>YEAR 1986</th>
<th>SEX AND AGE CLASSES</th>
<th>X1</th>
<th>mortal. Rate</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td>F</td>
<td>1500</td>
<td>0.04</td>
</tr>
<tr>
<td></td>
<td>C</td>
<td>1125</td>
<td>0.08</td>
</tr>
<tr>
<td></td>
<td>M1</td>
<td>675</td>
<td>0.01</td>
</tr>
<tr>
<td></td>
<td>M2</td>
<td>275</td>
<td>0.01</td>
</tr>
<tr>
<td></td>
<td>M3</td>
<td>150</td>
<td>0.05</td>
</tr>
<tr>
<td></td>
<td></td>
<td>3725</td>
<td></td>
</tr>
</tbody>
</table>
Fallow deer populations, as many other mammal species are not annual species, and moreover, before to shift into another age class, they stay one or more years in the same age class. It is reasonable above all if we consider the kind of reproductive behaviour they have to support. That’s the reason as we made a distinguish between age and stage classes: the first ones are consider one year exactly before shifting into another age; stage class models may enclose in one stage animals of different years depending on reproductive or mature condition of the species. In other terms, the basic assumption is that the demographic characteristics of individuals are related to their developmental stage.

If we now consider a model in which individuals in any class can either move to the next class or stay where they are, we could write a new diagram (Fig. 3.1), and observe a composite age class for individuals three-years-old and older with a final loop.

\[
L = \begin{pmatrix}
F_0 & F_1 & F_2 & F_3 \\
S_0 & 0 & 0 & 0 \\
0 & S_1 & 0 & 0 \\
0 & 0 & S_2 & S_{3+}
\end{pmatrix}
\]

in which: 
\[
S_{3+} = \frac{N_{4+}(t_2)}{N_3(t_1) + N_{4+}(t_1)}
\]

followed by the following diagram:

(Fig. 3.1 Diagram of an age-structured model with a composite age class for individuals three-years old and older)
Starting from the fallow deer population structure estimated in 1986 we calculated twenty Leslie Matrices to understand the behaviour of the population in this condition without any kind of harvesting. As we showed in a previous table, fallow deers have different age and sex classes, and to simplify the census we collected data only for five sex and age classes: adult female (all the females over 2 years; F); adult male (all the males over four years; M3); calve (the newborns, 0 – 1 year old; C); yearling (1 – 2 year old males; M1); pricket (2 – 4 year old males; M2). The difference in the number of female and male classes made necessary to build two different matrices.

The Leslie Matrix general forms for females and males, with each individual parameter calculated on the specific fallow deer population dynamics in San Rossore Estate, are shown:

\[
L = \begin{pmatrix}
0 & 0.75 \\
0.6 & 0.5 \\
\end{pmatrix}
\]

for females

\[
L = \begin{pmatrix}
0 & 0 & 0 & 0 \\
0.9 & 0 & 0 & 0 \\
0 & 0.9 & 0 & 0 \\
0 & 0 & 0.6 & 0.5 \\
\end{pmatrix}
\]

for males

The following matrix products show the first step of the Leslie Matrix running: we get results for the year 1987 starting from available data of year 1986.

\[
L = \begin{pmatrix}
0 & 0.75 \\
0.6 & 0.5 \\
\end{pmatrix}
\begin{pmatrix}
562 \\
1500 \\
\end{pmatrix}
= \begin{pmatrix}
1125 \\
1087 \\
\end{pmatrix}
\]

for females

\[
L = \begin{pmatrix}
0 & 0 & 0 & 0 \\
0.9 & 0 & 0 & 0 \\
0 & 0.9 & 0 & 0 \\
0 & 0 & 0.6 & 0.5 \\
\end{pmatrix}
\begin{pmatrix}
562 \\
150 \\
\end{pmatrix}
= \begin{pmatrix}
0 \\
506 \\
608 \\
240 \\
\end{pmatrix}
\]

for males
Twenty Leslie Matrix were developed to observe the behaviour of our fallow deer population under no culling control. Differently by other ungulate population living in no bounded areas (Festa-Bianchet, 2003) showing a first increasing in density, a plateau, and later a decreasing in number caused by reduction of natural resources, the fallow deer population in San Rossore Estate, avoiding from immigrant and emigrant flow, shows a slow decline depending on inbreeding depression (one of the first causes of diseases), overexploitation of trophic resources, spreading of parasites typical of overabundant population for the carrying capacity of the area. The youngest classes are the most hit classes. Poaching has only a secondary role on the population structure of the population. Obviously, it is not the aim of a Park, which consider this charismatic animal the symbol of the overall Conservation Site, driving this population to extinction destroying, at the same time, the environmental richness of the area.

Considering the proceeding in time of sex and age classes, what appears so strange and dangerous for the fitness of the population is the total alteration of the population structure: adult males result in a higher number respect females. Condition such that is no common in natural condition for wild animals, which use dispersal flows to restore the right proportion among sex and age classes. The following Figure (3.2) shows this unnatural condition.

(Fig. 3.2 Unnatural growth under no culling control programs between adult males (M3) and adult females (F) from 1986 until 2006).
The second step of the work is to implement the biological model on a bioeconomic one. What is important to consider, before developing a bioeconomic model, is the assumption that, in the biological model, as population size approaches carrying capacity, the number of fawns born per doe and survival rates of fawns declines, and natural mortality of adults increases.

We have yet specified in the previous section that the carrying capacity is defined as the maximum number of animals that the habitat can support, technically referred to as K-carrying capacity. At K-carrying capacity, there is no hunting mortality, and the number of deaths is exactly matched by the number of new individuals added to the population during the year. At each population size below K-carrying capacity, there is a number of animals that can be harvested annually (referred to as sustained yield) that will maintain that population size. The sustainable yield will equal the number of animals that would be added to the population if there were no hunting/harvesting activities. Guynn (1981) discussed in more details this argument. He explained the new harvest theory called “net present value”. This theory places a “present value” on a deer that is available for harvest, and compares that values to future ones as the deer increases in quality, but with a lower chance for harvest. In other terms, a buck that is alive and harvestable today, may be dead tomorrow or may never be seen again. Another theory of harvest management is the maximum sustained yield (Guynn, 1981). Under such a program, the total population is maintained through harvest at a certain level below carrying capacity. Anyway, these management plans are typically of hunting programs and, only seldom and in few countries, they can be used in Parks or other protected areas. The reason is that, while in a conservation site the main aim is to maintain or create a balance between all the parts acting (wildlife species and environment, monetary costs and benefits deriving by the management, as well as recreational benefits), in an hunting Estate the success of the program requires high reproduction, a stratified buck harvest and, in many hunting areas (e.g. in Scotland) antlerless deer are underculled to maximize profit deriving by trophy value. In conservation sites, as well as in our case study (San Rossore), the main reason to cull antlerless animals derives by the necessity to cull the highest number of animals, which will be able to give newborns ready to damage forestry and other environmental components. For an area like San Rossore Estate the main aim is to remove the higher number of
animals. Deers harvested alive are then sold to hunting Estate landowners, which will sell animals again to hunters and other stakeholders. To sell the higher number of animals San Rossore wildlife management policy set the same price both for males and females; in this way landowners are more attracted to buy females (the main problem for the Park), and at the same time, the get males at low price which will give an high profit in their own hunting Estate depending on their trophy value.

We postulate that Park officials seek to maximize the welfare provided by the deer herd to the all community (families, wildlife researchers, riders, walkers, photographers, etc.), by choosing a sequence of animal removals \( \{Y_t\} \) to solve an equation (Eq. 3.1) that seek to fit equal revenues and costs (like current laws in Italy provide for such like of management in protected areas) by fallow deer population management:

\[
p w Y_t - f(Y_t, X_t; \alpha, \beta) \times \text{wage/hours} = 0 \tag{3.1}
\]

where:

- \( p = \) is the price of shooted and harvested alive animals, obtained calculating the weighted mean;
- \( w = \) is the weight of animals in Kg, obtained calculating the weighted mean;
- \( f(Y_t, X_t; \alpha, \beta) = \) are the optimal hours calculated by Equation (A)

\[
h_{op} = \frac{Y_t - \alpha - b_0 X_t}{b_1} \tag{A}
\]

wage/hours = is the ranger wage per hour.

In such a way our aim is to find the optimal number of animals we can harvest \( (Y_t) \) from a given population \( (X_t) \) such that revenues and costs will be equal.

We later set up a multiple linear regression using six years data collected by Park Rangers, volunteers and researchers.

We calculated a multiple linear regression equation given by the following one (Eq. 3.2) using data from 2001 until 2006 to fit the coefficients \( (\alpha, b_0 \text{ and } b_1) \):

\[
Y_t = \alpha + b_0 X_t + b_1 \text{hours} \tag{3.2}
\]

Anyway the analysis was conducted for only one year, about which we collected more important biological information on rate of fertility, mortality and reproductive behaviour (year 2004).
The software SPSS 9.0 for Windows was used to develop the regression equation (Tab. 3.2), showing a linear relationship between the size of the culling and the number of hours of effort required for the harvesting.

In this way the regression relationship links the size of the cull to the population size and the number of hours of effort applied to the harvesting.

**Regression : results**

**Regression F Sig. Value**

*Sig. F* for the overall regression equation is estimated as following:

\[ \text{Sig. } F = 0.1 \]

*Sig. F* for the individual parameters of regression equation is estimated as following: \( \text{Sig. } F(\alpha) > 0.1 \)

\[ \text{Sig. } F(b_0) > 0.1 \]

\[ \text{Sig. } F(b_1) > 0.1 \]

CI : 95%

*(Tab. 3.2 SPSS results for the case of a Natural Park under law restrictions and no profit maximization)*

The following SPSS results show that the overall regression is almost statistically significant at the 10% level (\( \text{Sig. } F \) for the overall regression is \( 0.101 \)). The individual parameter estimates for \( \alpha, b_0 \) and \( b_1 \) are not statistically significant. However, with only 6 years of data it is unrealistic to expect a highly significant fit, but anyway these are the best results using only 6 years of data available to us.

Using a range of data collected over twenty or more years (as we will show for a case of profit maximization for a Park) suggest that the method used for our case study is correct, and the regression equation become highly statistically significant.

Using the values obtained by the SPSS 9.0 software and considering just one year for our analysis (the year 2004 for which we collected the most reasonable field data) we calculated the optimal number of hours, considering that for 2004 the Rangers culled a number of animals, \( Y_t = 1335 \), by a total population, \( X_t = 4001 \).
\[ h_{op} = \frac{Y_i - \alpha - b_0X_t}{b_i} = 7231 \] (optimal hours for an optimal harvesting). \hspace{1cm} (A)

In this way we obtained \( h_{op} \), which is just the number of hours of effort required to take a cull of size \( Y_i \) from a population of size \( X_t \). Obviously, \( h_{op} \) will change for any given cull size \( (Y_i) \) as the population size \( (X_t) \) changes during the years.

As we explained previously, the culling is divided in a number of animals harvested alive, and in a number of shot animals.

Considering that for the year 2004 the data collected by in the Park were like following:

**SHOT ANIMALS**

<table>
<thead>
<tr>
<th>Age</th>
<th>Number</th>
<th>Tot. Weight (Kg)</th>
<th>Price(€/Kg)</th>
</tr>
</thead>
<tbody>
<tr>
<td>1-2 year old males (M1)</td>
<td>29</td>
<td>1015</td>
<td>1.39</td>
</tr>
<tr>
<td>2-4 year old males (M2)</td>
<td>43</td>
<td>2150</td>
<td>1.39</td>
</tr>
<tr>
<td>Males over 4 years (M3)</td>
<td>0</td>
<td>0</td>
<td>1.39</td>
</tr>
<tr>
<td>Females over 2 years (F)</td>
<td>69</td>
<td>2691</td>
<td>1.39</td>
</tr>
<tr>
<td>Calves (C)</td>
<td>70</td>
<td>840</td>
<td>1.39</td>
</tr>
</tbody>
</table>

**HARVESTED ALIVE ANIMALS**

<table>
<thead>
<tr>
<th>Age</th>
<th>Number</th>
<th>Tot. Weight (Kg)</th>
<th>Price(€/Kg)</th>
</tr>
</thead>
<tbody>
<tr>
<td>1-2 year old males (M1)</td>
<td>251</td>
<td>8785</td>
<td>2.83</td>
</tr>
<tr>
<td>2-4 year old males (M2)</td>
<td>145</td>
<td>7250</td>
<td>2.83</td>
</tr>
<tr>
<td>Males over 4 years (M3)</td>
<td>45</td>
<td>3150</td>
<td>2.83</td>
</tr>
<tr>
<td>Females over 2 years (F)</td>
<td>351</td>
<td>13689</td>
<td>2.83</td>
</tr>
<tr>
<td>Calves (C)</td>
<td>332</td>
<td>3984</td>
<td>2.83</td>
</tr>
</tbody>
</table>

Weighted mean of the animal weights = 32.625 Kg
Weighted mean of the Price = 2.583 €
Ranger wage/hour = 20 €

From the equation (3.1)

\[ pwY_t - f(Y_t, X_t; \alpha, \beta) \times \text{wage} / \text{hours} = 0 \] \hspace{1cm} (3.1)
and using the optimal number of hours we obtained previously, we can calculate the optimal number of animals to be harvested (alive and shot), as we can observe in the following equation (Eq. 3.3)

\[ Y_t = \frac{h \times (\text{wage} / h)}{pw} = \frac{7231 \times 20}{2.583 \times 32.625} = 1716. \]  

(3.3)

So in year 2004 the optimal number of deers to be harvested is 381 animals more than the real number harvested by the rangers.

These results suggest that more animals could have been harvested in 2004 without incurring a financial loss (i.e. negative profit) from the harvesting operation. However, what is important to consider is that this conclusion is only valid if the regression relationship linking cull size, effort and population size is also valid. Our calculation suggests that it would have been possible to increase the cull by 29% in 2004 without incurring a financial loss from harvesting.

Considering confidence intervals in the regression model of 90%, we observed that the actual harvesting number obtained by the regression prediction fell inside the confidence intervals of 90%, suggesting that the method used is correct and that more animals should have been culled in that season.

Such kind of analysis calibrated on higher number of years, as the regression function requires (e.g., 20 years as we will set in the next case), suggests that this technique is correct showing high statistically significant results.

In a second approach we will try to analyse our Natural Park considering it like a private firm. So now the Park could be considered as a Sporting Estate with a private owner. Differently by the condition under law restrictions, as we showed previously, now the aim of the Park is to maximize the net present value of the net revenues from culling of a proportion of a fallow deer population (considered as a pest as well as a resource, and living in the Park boundaries where no natural predators are present), accounting for the venison and trophy value of culled animals and the costs of management (ranger’s wage; management of fenced captured sites; gasoline for cars; shotguns and bullets; costs incurred by damage to crops and forestry; etc).

A brief comment on this kind of bioeconomic approach to the management plan for a wildlife population could sound useful to understand the wildlife management
scenario in Italy. It is reasonable to think that in Italy, in a future perspective, laws regarding the management of Parks and other Protected Areas could change, depending on reduction of national funds for researches or for the management of critical or endangered species (considered both animals and plants). In other terms, any kind of a Protected Areas could become a private firm, able to achieve revenues for its management plans by its own activities to sustain better condition for environment and social welfare. Environmental education, excursions, horse-therapy, wildlife-therapy, bird-watch, biological production, and naturalistic researches, are only few of the activities that could be planned to reach a good profit at the end of the year, available to be used for sites of environmental importance and other management plans which need to be restored. At the same time, it means that the incoming of money from the culling of a proportion of a wildlife population, which cause damage to crops and forestry at high density, could become a consistent part of the final profit, above all if we desire to balance and reduce the financial loss due to these kind of damages. Thus, additional revenues got after these kinds of management plans, could then be invested in other actions.

Starting from a fixed population of fallow deers, our aim is now to obtain the number of culled deers (case A), or of hours of effort employed to harvest the animals by the rangers (case B), to have the possible maximum profit considered a population of deer whose number cannot be less than a minimum density ($X_{min}$) and not more than a maximum one ($X_{max}$). In the same way, the harvesting has to be calculated considered a minimum value and a maximum one ($Y_{min}$, $Y_{max}$).

Here we decided to analyze two cases: in the case (A) the number of hours of effort is calibrated on empirical calculation and by which we obtain the number of culled animals to achieve the maximum profit combined with the biological optimal number of animals in the population sustainable with the environment carrying capacity. The parameters of multiple linear regression equation (Eq. 3.2), that we obtained previously over six years of collected data, will be used to get the number of culling ($Y^*$), setting the number of hours of effort equal to 72 hours per ranger per deer management season, which will give the best profit available for the Park for the year 2004 using the new approach. In the second case (B) we used data from literature and simulation calculations on the population density and on the number of culling over twenty years.
Then, the parameters of the linear regression equation will set the optimal number of culled which now, in the multiple linear regression equation (3.8), will show the optimal hours of effort, necessary for our harvesting plan to obtain the maximum profit for the Park.

But let’s start step by step.

In this first case (Case A) we are going to calculate the maximum profit regarding the culling of a certain number of fallow deers starting from a known population \( X^*_t \) at year \( t \), and fixing the number of hours of effort by the rangers for the wildlife management plan. Mean marginal costs, which every year the Park have to sustain, are also set (costs for gasoline, for timber of the capture sites, for forage, for shotguns and boots, etc.), as well we know the ranger’s wage per hour.

The nomenclature for age and sex classes is shown as follows:

- \( F \) = over 2 year old females
- \( C \) = calves, 0-1 year old deers (also called fawns)
- \( M1 \) = yearlings, 1-2 year old males
- \( M2 \) = prickets, 2-4 year old males
- \( M3 \) = bucks, over 4 year old males

We consider again the equation (3.1) of this chapter and we call it Eq. (3.4) expressed as shown in the following form:

\[
\Pi^*_{\text{opt}} = \sum_{C}^{M3} (P_{i_{\text{deer-venison}}} \times Y_{i_{t}} \times P_{i_{\text{trophy}}} - f(Y_{i_{t}} , X_{i_{t}} ; \alpha , \beta ) \times (\text{wg/hour} \times \text{other costs}))
\]

(3.4)

where, for a fixed year, \( t \), we set the values in this way:

- \( X_{tot_{t}} = 1500 \), divided for age and sex classes as shown in Tab. 3.4
- \( Y_{i_{t}} \) = the size of the cull
- \( P_{i_{\text{deer-venison}}} \) = animal price for each age and sex classes referred to venison (€)
- \( P_{i_{\text{trophy}}} \) = trophy value (€) for male age classes
- Number of rangers employed for the wildlife management = 11
- Wg/hour (wage per ranger per hour) = 20 €
- Hours of effort per ranger per wildlife management season = 72 hours
Other costs for the wildlife management per season = 32000 €

\[ Wg/hour + Other \text{ costs} = \text{total costs} \]

In the following table (Tab. 3.3) we expressed the mean animal weight, the mean price of each animal depending on sex and age classes (deer value for venison), and the mean price for trophy (trophy value):

<table>
<thead>
<tr>
<th>Mean unit venison value for all age and sex classes (2.6 €)</th>
</tr>
</thead>
<tbody>
<tr>
<td><strong>Mean fallow deer weight (in Kg):</strong></td>
</tr>
<tr>
<td>F: 39 Kg</td>
</tr>
<tr>
<td>C: 12 Kg</td>
</tr>
<tr>
<td>M1: 35 Kg</td>
</tr>
<tr>
<td>M2: 50 Kg</td>
</tr>
<tr>
<td>M3: 70 Kg</td>
</tr>
</tbody>
</table>

<table>
<thead>
<tr>
<th>Mean fallow deer value (weight*unit venison price)</th>
<th>Mean trophy value:</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td></td>
</tr>
<tr>
<td>( P_{i, \text{deer-venison}} (€) )</td>
<td>( P_{i, \text{trophy}} (€) )</td>
</tr>
<tr>
<td>F: 100 €</td>
<td>0 €</td>
</tr>
<tr>
<td>C: 35 €</td>
<td>0 €</td>
</tr>
<tr>
<td>M1: 90 €</td>
<td>100 €</td>
</tr>
<tr>
<td>M2: 130 €</td>
<td>200 €</td>
</tr>
<tr>
<td>M3: 200 €</td>
<td>600 €</td>
</tr>
</tbody>
</table>

*(Tab.3.3 Mean animal weight, mean venison price and mean trophy price)*

The individual parameters of the regression equation (Eq. 3.2) were used in a new equation (Eq. 3.5) to obtain the optimal number of culling for this new simulating year, using the fixed initial population density (\(X_t^* = 1500\)) and the number of hours of effort (hours* = 792):

\[
Y_t^* = \alpha + b_0X_t^* + b_1\text{hours}^*
\]

From an initial population density of \(X_t^* = 1500\), a fixed number of hours of effort = 792, we obtained a harvesting \(Y_t^* = 853\), and a population density at time \(t+1\) as \(X_{t+1}^* = 1000\). This population considers the birth season, the mortality rate,
and obviously the shifting among age and sex classes. All results obtained are shown in the following table (Tab. 3.4):

<table>
<thead>
<tr>
<th></th>
<th>Xt</th>
<th>mortal. Rate</th>
<th>HARVESTING</th>
<th>Xt+1</th>
</tr>
</thead>
<tbody>
<tr>
<td>F</td>
<td>360</td>
<td>0.04</td>
<td>203</td>
<td>273</td>
</tr>
<tr>
<td>C</td>
<td>310</td>
<td>0.08</td>
<td>270</td>
<td>184</td>
</tr>
<tr>
<td>M1</td>
<td>230</td>
<td>0.01</td>
<td>180</td>
<td>123</td>
</tr>
<tr>
<td>M2</td>
<td>300</td>
<td>0.01</td>
<td>100</td>
<td>159</td>
</tr>
<tr>
<td>M3</td>
<td>300</td>
<td>0.05</td>
<td>100</td>
<td>261</td>
</tr>
<tr>
<td></td>
<td>1500</td>
<td></td>
<td>853</td>
<td>1000</td>
</tr>
</tbody>
</table>

*Tab. 3.4 Population structure before and later the harvesting plan for CASE A*

To expressed the equation (3.4) we will use the symbols showed in the following table (Tab 3.5):

<table>
<thead>
<tr>
<th><strong>IT</strong>&lt;sub&gt;opt&lt;/sub&gt;</th>
<th>the maximum profit available for the Park starting from a known initial deer population density, and known hours of effort employed for the culling management;</th>
</tr>
</thead>
<tbody>
<tr>
<td><strong>P&lt;sub&gt;F&lt;/sub&gt; deer-venison</strong></td>
<td>mean female price (considered only the venison value), in €</td>
</tr>
<tr>
<td><strong>P&lt;sub&gt;C&lt;/sub&gt; deer-venison</strong></td>
<td>mean calve price (considered only the venison value), in €</td>
</tr>
<tr>
<td><strong>P&lt;sub&gt;M1&lt;/sub&gt; deer-venison</strong></td>
<td>mean yearling price (considered only the venison value), in €</td>
</tr>
<tr>
<td><strong>P&lt;sub&gt;M2&lt;/sub&gt; deer-venison</strong></td>
<td>mean pricket price (considered only the venison value), in €</td>
</tr>
<tr>
<td><strong>P&lt;sub&gt;M3&lt;/sub&gt; deer-venison</strong></td>
<td>mean buck price (considered only the venison value), in €</td>
</tr>
<tr>
<td><strong>P&lt;sub&gt;F&lt;/sub&gt; trophy</strong></td>
<td>mean female trophy price, in €</td>
</tr>
<tr>
<td><strong>P&lt;sub&gt;C&lt;/sub&gt; trophy</strong></td>
<td>mean calve trophy price, in €</td>
</tr>
<tr>
<td><strong>P&lt;sub&gt;M1&lt;/sub&gt; trophy</strong></td>
<td>mean yearling trophy price, in €</td>
</tr>
<tr>
<td><strong>P&lt;sub&gt;M2&lt;/sub&gt; trophy</strong></td>
<td>mean pricket trophy price, in €</td>
</tr>
<tr>
<td><strong>P&lt;sub&gt;M3&lt;/sub&gt; trophy</strong></td>
<td>mean buck trophy price, in €</td>
</tr>
<tr>
<td><strong>Y*F</strong></td>
<td>size of the cull for the female class</td>
</tr>
<tr>
<td><strong>Y*C</strong></td>
<td>size of the cull for the calve class</td>
</tr>
<tr>
<td><strong>Y*M1</strong></td>
<td>size of the cull for the yearling class</td>
</tr>
<tr>
<td><strong>Y*M2</strong></td>
<td>size of the cull for the pricket class</td>
</tr>
<tr>
<td><strong>Y*M3</strong></td>
<td>size of the cull for the buck class</td>
</tr>
</tbody>
</table>

Wage, hours of effort and fixed costs of management are already set.

*Tab.3.5 Symbols adopted for equation 3.4*
We can now calculate the maximum possible profit for a Park, from a known initial population density and a number of hours of effort necessary for the wildlife management, obtaining the size of the cull by the regression mathematical calculation.

The equation (3.4) is now expressed in the following form:

\[
IP_{\text{opt}} = [(PF\text{ deer-venison } \times Y^*_{C}) + (PF\text{ trophy } \times Y^*_{C}) + (PC\text{ deer-venison } \times Y^*_{C}) + (PC\text{ trophy } \times Y^*_{C}) + (PM1\text{ deer-venison } \times Y^*_{M1}) + (PM1\text{ trophy } \times Y^*_{M1}) + (PM2\text{ deer-venison } \times Y^*_{M2}) + (PM2\text{ trophy } \times Y^*_{M2}) + (PM3\text{ deer-venison } \times Y^*_{M3}) + (PM3\text{ trophy } \times Y^*_{M3})] - [(\text{wg/ hours*}) + \text{other costs}]
\]

(3.4)

The results of the equation 3.4, together with the ones deriving by equation 3.8 are expressed in Table 3.8 for a better comparison between results obtained by a cull with a set up of the hours of effort or by a set up with the size of the cull.

The second case (Case B) is going to consider the condition by which we will to achieve the optimal available and sustainable profit for the Park, with a lack on number of hours of effort necessary for the culling. Starting from a progression over 20 years of simulating and calibrated data (see Tab. 4.5, Tab 4.6, Fig. 4.1). We are now considering a linear regression equation, in which \( Y_t^{**} \) and \( X_t^{**} \) are correlated to each other:

\[
Y_t^{**} = a_0 + b_0 X_t^{**}
\]

(3.6)

In this way, using data over twenty years, we obtained new individual parameters \((a_0, b_0)\) by which we are able to calculate the best culling referred on given initial population \(X_t^{**}\). Fig 3.3 show twenty years of harvesting plans, from 1986 until 2006.
The software SPSS 9.0 for Windows was used to solve the linear regression equation: the overall regression is highly statistically significant (Sig. $F$ for the overall regression is estimated as following: Sig. $F < 0.001$). The individual parameters estimates for $a_0$ and $b_0$ are still statistically significant (see Tab 3.6), as the overall regression, confirming that working over a long time-span gives us more information on the population structure subject to culling plans. The results obtained previously collecting real data only over six years showed a regression weakly statistically significant; but repeating the method over more years assure us that the methodology is correct, and so we can accept the result that more animals could have been harvested in 2004 without incurring a financial loss (i.e. negative profit) from the harvesting operation.

**Regression : results**

**Regression $F$ Sig. Value**

*Sig. $F$ for the overall regression equation is estimated as following:*

*Sig. $F < 0.0001$*
Sig. F for the individual parameters of the regression equation is estimated as following:

- Sig. F (a) < 0.0001
- Sig. F (b0) < 0.0001
- Sig. F (b1) < 0.0001

CI : 95%

(Tab. 3.6 SPSS results for the case of a Natural Park considered like a Sporting Estate aiming to maximize his profit and a lack of knowledge in number of hours of effort)

Finally, the last step is going to use the value of \( Y_{t}^{**} \) obtained by the Eq. (3.6) to solve a modified Eq. (3.2), now called equation (3.7), to have the optimal number of hours of effort employed for the optimal/sustainable profit (Eq. 3.8). Table 3.7 shows the population structure before and later the harvesting plan for CASE B).

<table>
<thead>
<tr>
<th></th>
<th>Xt</th>
<th>mortal. Rate</th>
<th>HARVESTING</th>
<th>Xt+1</th>
</tr>
</thead>
<tbody>
<tr>
<td>F</td>
<td>360</td>
<td>0.04</td>
<td>280</td>
<td>196</td>
</tr>
<tr>
<td>C</td>
<td>310</td>
<td>0.08</td>
<td>205</td>
<td>194</td>
</tr>
<tr>
<td>M1</td>
<td>230</td>
<td>0.01</td>
<td>200</td>
<td>107</td>
</tr>
<tr>
<td>M2</td>
<td>300</td>
<td>0.01</td>
<td>250</td>
<td>76</td>
</tr>
<tr>
<td>M3</td>
<td>300</td>
<td>0.05</td>
<td>250</td>
<td>70</td>
</tr>
<tr>
<td></td>
<td>1500</td>
<td>1185</td>
<td></td>
<td>643</td>
</tr>
</tbody>
</table>

(Tab.3.7 Population structure before and later the harvesting plan for CASE B)

Starting from the individual parameters obtained by SPSS 9.0 we calculated the optimal size of cull regarding a given initial population of fallow deer \( X_{t}^{**} = 1500 \):

\[
Y_{t}^{**} = a_0 + b_0 X_{t}^{**} = 1185
\]  

(3.6)

Combining this new harvesting value in the Eq. (3.2), equation now modified and called Eq. (3.7), we finally achieve the optimal number hours of effort necessary for the maximum sustainable profit (see Eq. 3.8).

\[
Y_{t}^{**} = a + b_0 X_{t}^{**} + b_1 \text{hours}^{**}
\]  

(3.7)

\[
\text{hours}^{**} = \frac{Y_{t}^{**} - a - b_0 X_{t}^{**}}{b_1} = 658
\]  

(3.7)
Using the number of hours of effort employed for wildlife management derived by Tab. 4.7, venison, trophy values and symbols used as shown in Tab. 4.3, we can now solve the last equation for the optimal profit available (Eq. 3.8)

$$\Pi^{\text{opt}} = [(P_F \text{deer-venison} \times Y^{\text{F}}) + (P_F \text{trophy} \times Y^{\text{F}}) + (P_C \text{deer-venison} \times Y^{\text{C}}) + (P_C \text{trophy} \times Y^{\text{C}}) + (P_{M1} \text{deer-venison} \times Y^{\text{M1}}) + (P_{M1} \text{trophy} \times Y^{\text{M1}}) + (P_{M2} \text{deer-venison} \times Y^{\text{M2}}) + (P_{M2} \text{trophy} \times Y^{\text{M2}}) + (P_{M3} \text{deer-venison} \times Y^{\text{M3}}) + (P_{M3} \text{trophy} \times Y^{\text{M3}})] - [(\text{wg} / \text{hours}) + \text{other costs}]] \quad (3.8)$$

The results of the equation 3.8 (as well as the ones of equation 3.4) are expressed in Table 3.8.

*(Tab. 3.8 Results by the equation (3.4) and (3.8))*

<table>
<thead>
<tr>
<th>With the number of hours fixed (CASE A):</th>
<th>With the size of the cull fixed (CASE B):</th>
</tr>
</thead>
<tbody>
<tr>
<td>$\Pi^*_{\text{opt}} = 129110 \text{ €}$</td>
<td>$\Pi^{**}_{\text{opt}} = 357950 \text{ €}$</td>
</tr>
</tbody>
</table>

**3.1.2 Conclusions**

Current Italian laws on wildlife management do not provide the possibility to achieve monetary benefits by wildlife population culling and selling. The main reason of such a law in Italy is that Italian Government policy does not want to reduce a Natural Area in a Sporting Estate where stakeholders (hunters) are authorized to kill animals as in many other hunting areas.

Anyway, if this law could be the right solution to the previous point of view, it is also true that a Natural Park could provide to manage the wild populations using Park Rangers, wildlife biologists and managers, specialized workers, and no local hunters. In this way monetary benefits deriving by the control of high wild population density could be used for other management action plans in the same Park. It is important to notice that this condition could drive to a new horizon for
Italian wildlife Park policy, above all considering that the current funding situation is characterized by a decreasing of National funds for environmental actions in Parks and other Natural Areas, and by common red tape conditions. So the possibility for a Park to manage itself could be the real future solution to take place.

This section aimed to show different approaches to the problem of controlling an overabundant fallow deer population in San Rossore Estate, fenced and bounded by the Tyrrehan sea, two rivers and a National roadway.

Leslie Matrices results over twenty years show the typical behaviour of an ungulate population under no culling control programs. The meaning of this assumption is based on a typical decreasing smoothly curve population density growth usually based, for ungulate population in wild habitat, on a first increasing in the number of animals, followed by a plateau condition, and a decreasing slope. That’s the typical growth curve for a wild ungulate population living in a no bounded area, so where natural conditions provided for dispersal flows and predation attacks. It is in these circumstances that emigration and immigration phenomena are common and, at the same time, necessary for the fitness of the same species. In San Rossore Estate, enclosed by two rivers, the sea and a national roadway, and in absence of natural predators, there is no the possibility of dispersal flows outside the reserve, avoiding the population to consume other kind of resources than the ones lying in the Park. Moreover, in these conditions, pathogen agents, easily to be spread among an enclose group, kill firstly the youngest classes, and the decreasing of the meaning trophic resources weigh primarily on the female class, which has to support birth, suckling and nursing (Casanova et al., 1988).

That’s the reason as it is not so surprising to notice that the number of adult males (class M3) increases in time while the number of adult females (class F) decreases proportionally over time (Fig. 3.5). In fact, we usually read more proportion of adult female class than the adult male one: the reason is strictly a biological condition as only a small proportion of adult males are able to breed with females (polygamous reproductive system), maintaining the percentage of adult females than males at least over 0.5%. In conditions such those, it is absolutely necessary the human role to equilibrate this lack of balance among all the environmental and the human parts taking in act.

In the first bioeconomic model, on the other hand we used results by a multiple linear regression over six years of data to calculate which could be the optimal
number of animals to be culled, in a condition in which the current Italian policy do not provide profit for the Park by the selling of animals: in other words, costs of management should equal revenues deriving by the selling. In the second case we assumed San Rossore Estate as a Sporting one in which the landowner will to maximize his profit by the culling considering a management plan over 20 years.

As we showed, in the first case the regression relationship links the size of the cull to the population size and the number of hours of effort applied to the harvesting. Although we used only six years of data to calculate which could have been the optimal number of deer to be culled without incurring a financial loss, we obtained results (supported by the same analysis over 20 years) showing that for a certain year (2004) it would have been possible to increase the cull by 29%, without incurring a financial loss from harvesting and without realizing a profit objective, maintaining a sustainable population density of the overall environment. To maintain a balance between revenues and costs deriving by harvesting plans, many animals are not sold but simply offered as donations.

In the second case we first supposed that San Rossore Estate is a Sporting Estate with a sole owner aiming to maximize his profit by the culling. Over 20 years of simulating data, we considered a first approach in which we fixed the number of hours of effort employed for wildlife management to obtain the optimal number of deer to be culled to achieve the maximum profit sustainable for the environment. In the second approach we fixed the culling; in this case the aim is to find the optimal number of hours of effort. The different results show how wildlife mangers could drive, in a financial and biological sustainable way, different profit functions or other environmental management plans necessary, year by year, to sustain the overall action plans in the Park.

What is interesting to notice in both the cases is that we decided to start the simulating approach from an initial population density of 1500 animals (meaning 32 animals/100 ha); in the first case (Case A) we set the number of hours of effort necessary for fallow deer management using the real number of hours employed by rangers for wildlife activities; we derived a sustainable number of animals to be culled equal to 853, reducing the population to 14 animals/100 ha. In the second case (Case B) we supposed that a wildlife manager fit the size of the cull at about 1185 animals for that season (meaning 18 animals/100 ha), obtaining the optimal number of hours effort by the regression relationship. In both the cases we got good
net revenues by the selling, allowable to be used for other management plans in the
same Area. Anyway, while both the approaches realize the aiming of the Park to
reduce the fallow deer population density to 12-14 animals/100 ha, the second
approach drive to a major monetary benefit. The main difference between the two
approaches is showed by the results: in fact, in the second case, when a specialized
biologist draw up the size of the cull, we can observe that the harvesting provide to
catch a good number of adult males (which means profit not only by venison value
but also by trophy value); at the same time the effort is reduced, allowing rangers or
other workers to be employed for other managing work in the Park. The final result
is that revenues by this last kind of culling program is able to get free the Park
policy by the current Italian “red tapes”, setting out own funds for other
management action plans. In other words, it is desirable a sustainable management
control program for wild pest animals to reduce the costs of the same management
and to maintain the balance among the environmental aspects taking in act.

3.2.1 Hunter’s utility maximization problem

One of the possible bioeconomic approach to a wildlife management plan could
be considered a work focused on deer (here we are not specifying which species of
deer we would consider as the method could be apply to many different species, and
it is not only valid for ungulates or herbivores) herd valuation, and its consequent
management in a Sporting Estate, or an area under hunting law permissions,
considering animals both as pests as resources. In the following case, for example,
we will consider relationship among herd dynamics, hunter utility, and the marginal
value of the herd, which are derived from the household production function
approach.

The population dynamics of the deer herd depend upon the physical
characteristic of the area, the weather, the presence of natural predators, the biology
of the habitat, and the hunter harvest.

At this point let the following equation

\[
\frac{dx}{dt} = f(x) - h
\]  

(a1)
summarize these relationships, where $h$ is the hunter harvest and $F(x)$ captures the effect of the other elements considered in the population dynamics. The quantification of population dynamics for the species under consideration is necessary for valuation of the marginal stock changes. We have yet observed in the previous chapter that several publications deal with mathematical models of population dynamics. The hunter harvest is the result of the interaction of utility-maximizing, price-taking hunters and the size of the population of deer in the area. The resulting harvest depends upon hunting laws and restrictions, tastes, prices, roads, technology, and obviously by the deer population level. Starting from a simple approach, and assuming that laws, tastes, and other variables are constants, we obtain the following equation:

$$h = h(x)$$  \hspace{1cm} (a2)

The population dynamics are captured in equations (a1) and (a2), and the initial herd size is given by $x_0$. Let the problem

$$\frac{dx}{dt} = f(x) - h(x)$$

have the solution

$$x(t) = g(x_0, t).$$  \hspace{1cm} (a3)

This equation identifies the time profile of the herd size for the initial herd size ($x_0$), reproduction rate, and the interaction of hunters and deers in the herd area.

In order to determine the value of a change in the stock of deer, we could define the aggregate benefits as a function of the stock of deer at any time $t$:

$$B = \int_0^\infty e^{-rt} s(x(t)) dt$$  \hspace{1cm} (a4)

where $e^{\cdot}$ is the natural exponential function (discount function), $r$ represents here the appropriate interest rate, $s(\cdot)$ is the aggregate compensating variation
consumers’ surplus function, \( x(t) \) is the herd size at time \( t \), and the starting time is zero. It is important to note that in this kind of approach the relationship between herd size and consumers’ surplus is implicit. The aggregate surplus function is

\[
s(x) = \sum_{j=1}^{n} s^j(x) \tag{a5}
\]

where, \( s^j(x) \) is the compensating variation consumers’ surplus for the \( j^{th} \) hunter, and \( n \) is the maximum number of hunters who hunt in the unit. At this step we decide to assume \( s^j(x) \) equals zero if a hunter does not hunt in the unit under the prevailing conditions.

Inserting equation (a3) into (a4) yields

\[
B^* = \int_0^\infty \exp(-rt)s(g(x_0,t))dt \tag{a6}
\]

The present value of the surplus stream now depends upon the starting population \( x_0 \), the biological growth rate, the interactions of hunters and deer \( (g(\cdot)) \), the aggregate consumers’ surplus function \( (s(\cdot)) \), and the interest rate. Differentiating \( B^* \) with respect to \( x_0 \) yields the shadow value of a deer in the initial population. This derivative is given by the following equation

\[
\frac{\partial B^*}{\partial x_0} = \int_0^\infty \exp(-rt) \frac{ds}{dx} \frac{\partial g}{\partial x_0} dt \tag{a7}
\]

where \( ds/dx \) is the marginal consumers’ surplus of the herd size at each point in time, and \( \partial g/\partial x_0 \) is the additional deer at each point in time causing additional deer in time zero. The derivative \( \partial g/\partial x_0 \) can be analyzed numerically or, for some \( f(x) \) and \( h(x) \) functions, solved analytically. The marginal consumers’ surplus, resulting from the change in deer population (stock), is determined by

\[
\frac{ds}{dx} = \sum_{j=1}^{n} \frac{ds^j(x)}{dx} \tag{a8}
\]
and the hunter’s utility maximization problem is:

Maximize \( U(Z) \) \hspace{1cm} (a9)

subject to:
\[
\begin{align*}
Z_1 &= F^1 (y^1, t^1) \\
Z_2 &= F^2 (y^2, r) \\
Z_3 &= F^3 (y^3, r, x) \\
Z_4 &= F^4 (y^4, r^t) \\
p \times (y^1 + y^2 + y^3 + y^4) - (b + Z_4w) &= 0 \\
t_1 + t_2 + t_3 + t_4 - T &= 0,
\end{align*}
\]

where \( Z_1 \) is a composite commodity, \( Z_2 \) is the quantity aspect of hunting, \( Z_3 \) is the quality aspect (which is assumed to be hunter success), \( Z_4 \) is the hours of week, \( F^i (\cdot) \) are the household production functions, \( y^i \) are vectors of purchase goods, \( p \) is a vector of good prices, \( t^i \) is time spent producing \( Z_i \), \( T \) is the total quantity time in the time period, \( b \) is non-labor income, and \( w \) is the wage rate.

In order to generate an expression for \( ds^j(x)/d(x) \) (expressed by the equations \( a7 \) and \( a8 \)), the dual problem is raised by:

\[
\text{minimize : } b = p \times (y^1 + y^2 + y^3 + y^4) - Z_4w \hspace{1cm} (a10)
\]

subject to the time and commodity production constraints in (a9). The result of this kind of approach is the compensated non-labor income function.

Differentiation of this solution function \( (b^*) \) using the envelope theorem yields:

\[
\frac{\partial b^*}{\partial x} = \xi_3 \frac{\partial F^3}{\partial x} \hspace{1cm} (a11)
\]

where
\( \xi, \) is the Lagrangean multiplier for \( F^3(\cdot) - Z_3 = 0. \) Further, it can be shown that this derivative is the negative of the derivative of compensating variation consumers’ surplus:

\[
\frac{ds^j(x)}{dx} = -\frac{\partial b^*}{\partial x} = -\xi_3 \frac{\partial F^3}{\partial x} \tag{a12}
\]

In this formulation, \( -\xi_3 \) is the shadow value of hunter success \( (Z_3), \) and \( \partial F^3/\partial x \) is the change in hunter success with a change in herd size. Thus:

\[
\frac{ds^j(x)}{dx} = (\text{shadow value of hunter success}) \ast (\text{marginal responsiveness of hunter success to herd size}) \tag{a12'}
\]

Equation (a12’) indicates that for the \( j^{th} \) hunter, the marginal value of deer in the herd is the product of the shadow value of the quality variable (hunter success) and the marginal responsiveness of hunter success size. Equation (a18) indicates that the individual shadow values are summed to obtain the aggregate shadow value of deer at a point in time. Finally, the shadow value of an additional deer in time zero is identified by the integral in equation (a7) to be discounted, aggregate shadow value of the stream of current and future effects of this additional deer.

### 3.2.2 Conclusions

This first example tried to explain the hunter’s utility maximization problem throughout the consideration that much of the recreation valuation information on which resource managers base allocations is inappropriate, particularly for wildlife management. Economics have been providing wildlife managers with recreational values that are not appropriate for most management decisions. The values currently applied to publicly-provided wildlife-related recreation have been based on estimates of average visitation or harvest values in inframarginal contexts. Marginal analysis of management practices has generally been ignored, as the difference between the value of a harvested animal and benefits and costs of adding to the reproducing stock (Batie and Shabman, 1979). For example, Sorg and Loomis (1984) reported 15 valuation studies on big-game hunting, all of which were based
upon visitor-days, as opposed to valuing the wildlife directly. Although a relationship among visitor-days, hunter success, and big-game populations could have been estimated, the value generated in those studies (consumer’s surplus per visitor-day) fail to provide sufficient information for management decisions. Moreover, since managers generally are limited to either habitat manipulation or harvest constraints, it is necessary to focus valuation on the marginal value of these efforts. This requires analyzing the marginal value of changes in the stock of wildlife, rather than an average value of visitor days. Furthermore, optimal management must involve considering the effectiveness of habitat manipulation, or other controls, on the stock of wildlife as well as the cost of these practices. Given that public agencies will likely continue to be the major provider of public recreation activities, it is essential that biologists and economists cooperate in research will lead to the appropriate information being collected and analyzed in a theoretically correct way.

3.3.1 Wildlife management maximizing the social welfare

An other possible starting point could be the one establishing a link between the renewable resource and pest control literature by extending Jacquette’s results (1972; 1974) to situations in which stock growth is deterministic, non-linear and stock dependent, where the animal population generates both social costs and benefits. In this new case we could identify a dynamic cost effect, resembling the dynamics at work in many other fishery studies, as the primary cause for the efficiency gains in pulsing. Usually this kind of approach provides a first step in which a basic deterministic model of our wild population is developed, and a second step in which we calibrate the model.

By seeking the advise of community members, states, counties, and town officials sought to balance the damage caused by deer over-abundance against the recreational and other non-consumptive benefits of deer (e.g. wildlife observation, amenity value) and the costs of controlling the population. We therefore postulate that town (as well as district, or regional, or national) officials seek to maximize the welfare provided by deer herd to the community, by choosing a sequence of animal removals ($Y_t$) to solve
Maximize \[ \sum_{t=0}^{n} \rho^t \left[ \frac{B(X_{t-1})}{\rho} + \left( p - \frac{c}{X_{t-1}} \right) Y_t - SC \right] \] 

Subject to

\[ X_t - X_{t-1} = F(X_{t-1}) - Y_t \left[ 1 + \frac{F(X_{t-1})}{X_{t-1}} \right] \]

\[ Y_t \leq \alpha X_{t-1} \]

\[ SC = \begin{cases} 0 & \text{if } Y_t = 0 \\ q > 0 & \text{if } Y_t > 0 \end{cases} \]

where

\[ SC = 0 \quad \text{if } Y_t = 0 \]
\[ SC = q \quad \text{if } Y_t > 0 \]

\( X_{t-1} \) given

In the equation (b1) \( t \) is an index of time denoting years, \( X_t \) is the number of animals in the population at year \( t \), \( Y_t \) is the number of animals removed in a culling operation at year \( t \), \( \rho \) is the annual discount factor, \( p \) is the value derived from consuming the venison (deer meat) of culled deer, \( \alpha \) is the proportion of the herd \( X_t \) that can physically be removed in year \( t \), \( SC \) is the set-up costs if management actions are taken in year \( t \), \( SC \) is the net benefit derived from population \( X_t \) in year \( t \).

It is assumed that \( B(X) \) is a strictly concave, single-peaked function reaching its maximum at \( X = \overline{X} \) with \( B_x > 0 \) for \( X \in [0, \overline{X}) \), \( B_x(\overline{X}) = 0 \) and \( B_x < 0 \) for \( X \in (\overline{X}, \infty) \); \( c \) is a variable cost parameter, \( F(X_{t-1}) \) represents the net change in the animal population occurring without deliberate human intervention between periods \( t-1 \) and \( t \).

The particular timing of the events adopted for this model reflects the typical approach to deer culling. The better population estimate is in autumn or early winter (so after the births, which occur in May-June), after which the culling takes place.
Since a proportion of the animals removed during the winter are pregnant females (the breeding season for deers is in autumn), the effective impact of culling on herd’s population in the next period is assumed equal to

\[ Y_t[1 + F(X_{t-1})/X_{t-1}] \]

Note that as \( F(X) \) measures annual mortality (for instance, high mortality of young fawns) as well as fecundity, this assumption slightly underestimates the number of embryos taken by the cull and the effectiveness of winter hunting. Thus the net change in the number of deers between year \( t-1 \) and \( t \) is given by subtracting this “effective take” from the natural growth \( F(X_{t-1}) \). The resulting population generates benefits and damages measured by \( B(X_t) \). The function \( B(X) \) is a parabola that measures both the positive benefits and the negative costs of deer herd. For a stock lower than \( \bar{X} \), the net benefits from the stock are increasing, whereas for stocks exceeding this critical value they are decreasing. Because culling operations can cover only part of the territory, a constraint exists on the number of animals that can be taken in a given year. For simplicity it is assumed that a constant proportion (\( \alpha \)) of the stock resides in and around the areas where culling takes place. Moreover, because the linearity of the cost function, the model admits the possibility of a “bang-bang” control solution whereby the wildlife manager adjusts the stock as rapidly as possible to bring it to its eventually steady state.

It can be shown that the steady state of the “bang-bang” control for the unconstrained problem (\( \alpha = 1 \)) is the stock \( X \) that solves the following equation:

\[
\delta = F_x - \left( \frac{F(X)}{X + F(X)} \right) \left( \frac{X F_x(X) - F(X)}{X} \right) + \frac{B_s}{p} \left( \frac{X + F(X)}{(p-c/X)X} \right) + \frac{c F(X)}{X^2 (p-c/X)}
\]

(b2)

Unfortunately, the presence of set-up costs and stock-dependent variable culling costs violates the sufficient conditions for the optimality of “bang-bang” solutions. For more details on this problem occurring in this kind of approach to a wildlife management action see Spence and Starret (1975).
3.3.2 Conclusions

With this second example reported, we aimed to analyze wildlife management maximizing the social welfare in a theoretically way. The results suggest that managing urban animal populations for a steady state can lead to inefficiencies and reduced community welfare. When constrained by the urban setting in which animal control activities must take place, the optimal management of a district (or a county or an urban Park) deer herd dictates that authorities harvest as many animals as safety constraints will allow, when (and only when) the stock exceeds an endogenously determined threshold level $X^*$. This constrained policy is a specific case of the unconstrained ($x, X^*$) regime in which the animal population is immediately reduced to $x$ whenever it exceeds $X^*$. This result was shown to be optimal for a wide range of policy-relevant parameter values. It extends Jacquette’s results (1974) to situations where the animal population provides benefits in addition to imposing damage, and to cases where nonlinear and stock-dependent growth is either deterministic or disturbed by random shocks. The negative impact of culling on future harvesting costs is the primary cause of optimal pulsing. In wildlife management situations where a herd is confined and a significant portion of the stock can be removed in a single season, the dynamic cost effect can easily create intertemporal economies of scale making pulsing optimal. For larger resource pools such as fisheries, pulsing has been historically viewed as a technical curiosity (Clark, 1990).

The ability of wildlife managers to harvest a large proportion (if not all) of a well defined wildlife population confined to an urban area or a specific and accessible Natural Area, the presence of set-up costs, and an inverse relationship between the population level and the marginal cost of harvesting are the central factors making pulsing superior to traditional steady-state management. What we can understand is that pulsing is less likely to be optimal in situations where fixed costs are low (conferring little economies of scale), or when harvesting instantaneously results in an increase in marginal harvesting costs (making little difference whether the next animal is taken from a population that was high or low at the beginning of the culling period). Therefore, pulsing is less likely to offer advantages when animals are elusive and cannot easily and effectively be rounded up or baited (e.g. coyote, alligator, raccoon). On the other hand, it is more likely to be efficient when the
animals are in a well-defined or confined area, and where fixed set-up or disutility costs are a significant proportion of total control costs.

### 3.4.1 Wildlife management maximizing meat and trophy value in ecological equilibrium

The aim of this further approach is to demonstrate the economic content of a structured wildlife population model and, later, to show how its economic contents may change under different management scenarios. Our starting point will be a typical Scandinavian management plan (Skonohoft and Olaussen, 2005), where wildlife animals (moose, red deers, wolves, etc.) are an important economic source (think about the increasing of hunting tourism in these countries, for example); at the same time they are valued as pests when their high densities run into human activities. As in a typical bioeconomic approach, we start formulating a population model for our population, for example moose (*Alces alces*), but it is a management action applicable to many other wild species. In a second part, we will try to demonstrate what happens when the hunting is steered by the traditional landowner goal of maximizing meat value. Next, we will try to analyze the sex and age composition under the new exploitation regime of trophy hunting. Finally, we will use some numerical illustrations from literature (Tab 3.9 and 3.10).

We decided to consider a moose population characterized by non-harvest mortality rates, generally low, due to lack of predators, and with no evidence of density-dependent mortality. From literature on moose and other ungulates in Europe, fecundity has proven to be affected by the female density, while the number of males seems to be of negligible importance (see Nilsen et al., 2005, for more details).

In our new empirical case we will consider the moose population structured in four stages: calves, yearlings, adult males and adult females. In what follows, however, to obtain a traceable analytical model, while still catching the main ecological content, just three stages are considered as yearlings are left out. The population at time (year) $t$ is hence structured as calves $X^0_t$, adult females $X^f_t \ (\geq 1 \ year)$, and adult males $X^m_t \ (\geq 1 \ year)$, so that the total population is expressed by $X_t = X^0_t + X^f_t + X^m_t$. Yearlings are therefore included among adult male and female classes. These three stages are henceforth called young, female and male. Usually the population density is measured from spring to summer. Moose hunting, in
Scandinavian countries, takes place in September – October. All natural mortality is assumed to take place during winter, after hunting season, as the natural mortality throughout summer and fall is small and negligible, considering also low predator densities. We decided to impose the same natural mortality rate for males and females. As we said previously, natural mortality is fixed and density independent, while reproduction is density dependent. It is further assumed the same sex ratio for young when they enter the old stages (Nilsen et al., 2005).

Neglecting any stochastic variation in biology and environment, and neglecting any dispersal in and out of the considered area, the young at time \( t+1 \) is first governed by:

\[
X_0^{t+1} = r_t X_f^t \tag{c1}
\]

where \( r_t > 0 \) is the fertility rate (number of young per female). The fertility rate is dependent on the female density:

\[
r_t = r(X_f^t) \tag{c2}
\]

with \( \frac{dr}{dX_f} = r' \) (when omitting the time subscript) and where \( r(0) > 0 \) is fixed.

Combining equation \((c1)\) and \((c2)\) gives the recruitment function \( X_0 = r(X_f^t) X_f^t \) with \( \frac{dX_0}{dX_f} = (r'X_f^t + r) \). The recruitment function is assumed to be concave. For obvious reasons \( \frac{dX_0}{dX_f} \geq 0 \) should be hold in an optimal harvesting programme.

The abundance of (old) female follows next as:

\[
X_f^{t+1} = 0.5(1 - m_0^t) (1 - h_0^t) X_0^t + (1 - m)(1 - h_f^t) X_f^t \tag{c3}
\]

where \( m_0^t > 0 \) and \( m > 0 \) are density independent mortality fractions of youngs and females and males, respectively, while \( h_0^t \) and \( h_f^t \) are the harvesting fractions. Half of the young population enters the females, after harvesting and natural mortality. The number of (adult) males is finally given by:

\[
X_m^{t+1} = 0.5(1 - m_0^t) (1 - h_0^t) X_0^t + (1 - m)(1 - h_m^t) X_m^t \tag{c4}
\]

where \( h_m^t \) is the male harvesting fraction.

When combining equation \((c1) - (c3)\), the female population dynamic reads:

\[
X_f^{t+1} = 0.5(1 - m_0^t) (1 - h_0^t) r(X_f^t) X_f^t + (1 - m)(1 - h_f^t) X_f^t \tag{c5}
\]

It is a second order non–linear difference equation, and numerical analyses demonstrates that the equilibrium is stable for fixed harvesting fractions (see Gandolfo, 2001, for a theoretical exposition). Omitting the time subscript, the equilibrium reads:

\[
X_f = 0.5(1 - m_0) (1 - h_0) r(X_f) X_f + (1 - m)(1 - h_f) X_f \tag{c5}
\]
There are two equilibria; the trivial one of \( \dot{X} = 0 \) and \( X > 0 \) is given by

\[
1 = 0.5(1 - m^0)(1 - h^0)r(\dot{X}) + (1 - m)(1 - h^0). \]

Consider that \( r' < 0 \), the non-trivial equilibrium will be unique and may be written as:

\[
\dot{X} = F(h^0, h^f) \tag{c5'}
\]

where \( F(\cdot) \) represents a functional form. The solution is an equation such that

\[
\partial F / \partial h^0 = F_0 < 0 \text{ and } F_f < 0.
\]

The iso-population female lines slope therefore downwards in the \((h^0, h^f)\) plane, and where the lines closer to the origin yield a higher stock.

In the same way, by combining the equations (c1), (c2) and (c4), the male population growth as the following equation reads:

\[
X_{m,t+1} = 0.5(1 - m^0)r(\dot{X}_{t})X_{f,t-1} + (1 - m)(1 - h^m)X_{m,t}. \]

The dynamic of the males is therefore contingent upon the female growth (but not vice versa as only female abundance regulates fertility), and again numerical analyses demonstrate that the equilibrium is stable. The equilibrium is:

\[
X_m = 0.5(1 - m^0)(1 - h^0)r(\dot{X})X_f + (1 - m)(1 - h^m)X_m \tag{c6}
\]

There are two equilibria for the male population as well: the trivial one is when \( X_f = 0 \), and \( X_m > 0 \) when \( X_f > 0 \). Equation (c6) may also be written as:

\[
X_m = G(h^0, h^m)r(\dot{X})X_f \tag{c6'}
\]

where \( G(h^0, h^m) = 0.5(1 - m^0)(1 - h^0) / [1 - (1 - m)(1 - h^m)] \). Again it may be confirmed that higher harvesting rate mean fewer animals, \( G_0 < 0 \) and \( G_m < 0 \). The male iso-population lines hence slope downwards in the \((h^0, h^m)\) plane, and lines closer to the origin yield a higher stock. On the other hand, a higher female subpopulation shifts these iso-population lines away from the origin (suggested that the slope of recruitment function is positive, as we explained above) meaning that there is place for more male harvesting for a given young sub-population harvesting, and vice versa.

Equation (c6') also indicates that the equilibrium male-female proportion decreases with more females. However, the male-female proportion may be more easily recognized when combining equation (c5) and (c6) which yields

\[
X_m / X_f = [1 - (1 - m)(1 - h^f)] / [1 - (1 - m)(1 - h^m)].
\]

We therefore simply have \( X_m / X_f = 1 \) if \( h^m = h^f \), as the mortality of the male and female are equal, and the same fraction of young enters the female and male subpopulations. The traditional exploitation of Scandinavian moose population, as
many other ungulate populations in Europe under a hunting regime, is direct by the maximization of the meat value in ecological equilibrium. Because natural mortality takes place after the hunting season, the equilibrium number of animals removed is simply $H^0 = h^0rX^f$, $H^f = hr^FX^f$ and $H^m = h^mX^m$ so that the total harvest equals $H = H^0 + H^f + H^m$. Accordingly, the management goal of the landowner is to

$$\max_{X^f, X^m, h^0, h^f, h^m} U = p\left[w^0H^0 + w^fH^f + w^mH^m\right] = p\left[w^0h^0r(X^f)X^f + w^fh^fX^f + w^mh^mX^m\right]$$

subject to the ecological constraints (c5') and (c6'). Note that $w^0 < w^f < w^m$ are the (average) body slaughter weights (Kg per animal) of the three stages, while $p$ is the meat price (€/Kg, for example, depending on the country). However, the meat price will not affect the optimization, except for scaling the shadow price values, as we show below.

The Lagrangian of this problem writes

$$L = p\left[w^0h^0r(X^f)X^f + w^fh^fX^f + w^mh^mX^m\right] - \lambda[X^f - F(h^0, h^f)] - \mu[X^m - G(h^0, h^m)r(X^f)]$$

with $\lambda \geq 0$ and $\mu \geq 0$ as the shadow prices of the female and male population respectively. Briefly, we would like to explain that the interpretation of $\lambda$ and $\mu$ as shadow prices is not obvious as the population sizes are determined within the model. However, when adding $\overline{X}^f$, interpreted as an exogenous number of introduced females to the stock as expressed in equation (c5'), it can be shown that $\partial U^\ast / \partial \overline{X}^f = \lambda$, and where $U^\ast$ denotes the maximum value of $U$. In the same manner, adding $\overline{X}^m$ as an exogenous number of introduced males to the equation (c6'), gives $\partial U^\ast / \partial \overline{X}^m = \mu$.

The first order conditions of this maximizing problem are (note that the second order conditions are fulfilled due the concavity of the recruitment function):

$$\partial L / \partial X^f = p\left[w^0h^0(rX^f + r) + w^fh^f\right] - \lambda + \mu G(rX^f + r) = 0$$
\[ \frac{\partial L}{\partial X^m} = pw^m h^m - \mu = 0 \quad \text{(c9)} \]

\[ \frac{\partial L}{\partial h^0} = pw^0 r X^f - \lambda F^0 + \mu G_0 r X^f \leq 0; \quad 0 \leq h^0 < 1, \quad \text{(c10)} \]

\[ \frac{\partial L}{\partial h^f} = pw^f X^f + \lambda F_f \leq 0; \quad 0 \leq h^f < 1, \quad \text{(c11)} \]

and

\[ \frac{\partial L}{\partial h^m} = pw^m X^m + \mu G_m \geq 0; \quad 0 < h^m \leq 1 \quad \text{(c12)} \]

Conditions (c8) and (c9) steer the shadow price value, and equation (c9) say that the male shadow price should be equal its marginal harvesting value. Equation (c8) is somewhat more complex, but indicates that the female shadow price should be equal the sum of the marginal harvesting value of the female and the young sub-populations, plus the indirect male marginal harvesting value, evaluated as its shadow price. Rewriting the equation (c8) using the conditions of equation (c9) we obtain

\[ \lambda = p(w^0 h^0 + w^m h^m G)(r X^f + r) + pw^f h^f. \]

As the slope of the recruitment function is non-negative, we have that \((r X^f + r) \geq 0\) and \(\lambda \geq pw^f h^f\). Hence, while the shadow value of the male population is exactly equal its marginal harvesting value, the shadow value of the female population is above its marginal harvesting value. In this sense, females may be considered as more “valuable” than males, in line with biological notion of females as valuable and males as non-valuable. Conditions (c10) and (c12) are the control conditions with the actual complementary slackness. From the male control condition (c12), harvesting down the whole population is hence considered as a possibility as this is the biological “end” product. On the other hand, keeping the female and young sub-population unexploited are also options as these stages represent the reproductive and potentially reproductive biological capital. Condition (c10) indicates that the harvesting of young should take place upon the point where the harvesting benefit is equal, or below, the cost in terms of reduced population of males and females.
evaluated at their respective shadow prices. When condition \((c10)\) holds as an inequality, the marginal harvesting benefit is below its marginal cost and harvesting is thus not profitable, \(h^0 = 0\). The interpretation of the female harvesting condition \((c11)\) is somewhat simpler. Due to the fecundity density effect, meaning that one more female on the margin yields a smaller recruitment when the female population is “high” than when being “low”, \(h^f = 0\) seems less likely.

The male harvesting condition \((c12)\) is analogue the female harvesting condition \((c11)\), but the cost-benefit ratio generally works in opposite direction. The condition always holds as an inequality. This is revealed when first combining condition \((c9)\), \((c12)\) and \((c6')\) which yields

\[
(G + h^m G_m) \geq 0.
\]

When next inserting for \(G\) (and \(G_m\)) from equation \((c6)\), we find that

\[
0.5(1 - m^0)(1 - h^0)/[1 - (1 - m)(1 - h^m)] \cdot [1 - (1 - m)h^m/[1 - (1 - m)(1 - h^m)] \geq 0.
\]

After some small rearrangements, we can note that it boils down to \(m \geq 0\). Accordingly, because \(m > 0\), \(h^m = 1\) and the whole male population should be harvested. Notice that this value holds irrespective of the meat value of males and females (as given by body weights).

The reason for harvesting down the whole biological “end” product as the best option is the lack of any trade-offs when the meat value is maximized; there is neither any biological feedback effects from the other stages nor any price demand response. The male-female proportion becomes accordingly

\[
Xm/Xf = [1 - (1 - m)(1 - h^f)]
\]

in the optimal program while one more male yields a benefit of \(\mu = pw^m\) (€/animal, for example).

If the optimal harvesting policy at the same time gives \(h^0 = 0\), the female shadow price reads

\[
\lambda = pw^m G(r'Xf + r) + pwfh^f.
\]

As \(G = 0.5(1 - m^0)\) when \(h^m = 1\) and \(h^0 = 0\) (equation \(c6')\) and \(wf < w^m\), the female shadow price may be lower than the male shadow price in contrast to the above notion of female as more “valuable” than males. In addition, from condition \((c8)\), it may also be shown that if
\[1 - G(r'X_f + r) > w'f', \text{ then } \mu > \lambda.\] This shows more directly that a low female slaughter weight may pull in the same direction.

Let’s try now to consider wildlife population harvesting under a trophy-hunting regime. In Scandinavia, as in many other countries in Europe (see Anderson and Hill, 1995), the moose, or other ungulate harvesting regimes, are gradually changing, and a hunting and wildlife industry, as well as a new kind of tourism, based on trophy value, is increasing in importance. “Modern times” are modelled by introducing a market for trophy hunting males, while still having meat value hunting for the other stages. The market for trophy hunting is probably something between a competitive market and monopoly. One of these extremes is chosen, and we assume that trophy hunting licences are supplied under monopolistic conditions. Following the common practices for trophy hunting, it is assumed that one licence allows the buyer to kill one selected animal (selected on previous studies on population dynamics and structure), which is paid only if the animal is killed. In some countries, stakeholders pay for a licence not depending on the killing of the animal. In addition to price, the demand for trophy hunting licences may also be contingent upon “quality”, expressed by the abundance of males.

The inverse market demand for male hunting licences is hence given as:
\[q = q(h^mX^m, X^m)\]  
(c13)

The licence price \(q\) (€/animal, for example) decrease with a higher offtake,
\[q_H = \frac{\partial q}{\partial (h^mX^m)} < 0,\] while it shifts up with more animals available, \(q_X > 0\).

Supplying trophy hunting licences is also costly and depends on the number of animals shot:
\[C = C(h^mX^m)\]  
(c14)

with the fixed cost \(C(0) > 0,\) and variable cost \(C' > 0.\) The fixed component includes the cost of preparing and marketing the hunting, whereas the variable component includes the cost of organising the permit sale, the costs of guiding and various transportation services, and so forth. The landowner management goal is now accordingly to find a harvesting policy that maximizes the sum of the meat value and trophy hunting profit:
\[
\max_{X^m, h^m, x^f, h^f} \pi = p[w^0 h^0 r(X^f) X^f + w^f h^f X^f] + [q(h^m X^m, X^m) h^m X^m - C(h^m X^m)]
\]

(c15)

again subject to the constraints \((c5')\) and \((c6')\). The first order conditions of this problem are (where \(L\) refers to the Lagrange function):

\[
\frac{\partial L}{\partial X^m} = q h^m X^m + q h^m - C' h^m + q_x h^m X^m - P = 0 \quad (c16)
\]

and

\[
\frac{\partial L}{\partial h^m} = q h^m X^m + q X^m - C' h^m + \mu G_m X^m \geq 0; \quad 0 < h^m \leq 1, \quad (c17)
\]

in addition to conditions \((c8), (c10)\) and \((c11)\).

The male harvesting benefit is now expressed by a marginal profit term plus a marginal stock effect through the demand quality effect, and the interpretation of \((c16)\) and \((c17)\) is straightforward. Combining these conditions with equation \((c6')\) yields

\[
(q h^m X^m + q - C')(G + h^m G_m) q_x h^m X^m G_m \geq 0,
\]

where the term \((G + h^m G_m)\) is still strictly positive because \(m > 0\). When first disregarding the quality effect, \(q_x = 0\), not harvesting all males, \(h^m < 1\), will hence represent the optimal solution if the marginal harvesting profit is equal to zero, \((q h^m X^m + q - C') = 0\). From condition \((c16)\), as well as from equation \((c17)\), we observed that this implies a zero value male shadow price. A zero value shadow price while not harvesting down the whole is a counterintuitive result, but hinges on the biological “end” product nature of the adult males; the number of males does not affect fertility. The zero marginal harvesting profit condition may be met if the marginal cost is high and/or the inverse demand schedule is steep (in-elastic). On the other hand, if the marginal revenue exceeds the marginal cost for \(h^m = 1\), we obtained the same solution type as well as \(\mu\) is positive.

When taking the demand quality effect into account, \(q_x > 0\), \(h^m < 1\) may still hold as an optimal solution when the marginal revenue exceeds the marginal cost \((q h^m X^m + q - C') > 0\), as \(G' < 0\) and \((G + h^m G_m) > 0\). The economic reason for this result is simple as constraining the harvest and keeping a high stock size works in the direction of a higher trophy hunting licence price through the quality shift in demand. From equation \((c16)\), it is seen that this situation implies \(\mu > 0\). The corner
solution of \( h^m = 1 \) is also now a possibility, but the marginal harvesting profit must then exceed a certain minimum, equal the shadow price.

While the first order conditions for harvesting female and young are the same as in the traditional harvesting regime, the new conditions for male harvesting will obviously spill over to these stages. If we consider \( h_m < 1 \), we may typically find that the male-female proportion \( X^m/X^f \) increases compared to the traditional regime which may be reinforced if \( h^H \) shift up at the same time. Moreover, while the meat price \( p \) had no effect on the optimal harvesting policy in the traditional regime, it may now influence the optimal harvesting policy of all the sex and age classes. This will generally occur when the quality effect is included and we have \( \mu > 0 \). In line with standard harvesting theory, more harvest and fewer animals of male and young is accompanied by a higher price. On the other hand, with no quality effect and with a zero shadow price value of the males, \( p \) will have no effect, and the conditions \((c5),(c8),(c10)\) and \((c11)\) will determine \( h^0, h^H, X^f \) and \( \lambda \).

We can now try to illustrate numerically the two exploitation schemes using data from literature (Nilsen et al., 2005; Storaas et al., 2001). The fecundity rate, decreasing in the number of females, is specified as a sigmoidal function with an increasing degree of density dependence at high density:

\[
    r = r(X^f) = \frac{\hat{r}}{1 + (X^f/K)^b}
\]

with \( \hat{r} > 0 \) as the intrinsic growth rate (maximum number of young per female) and \( K > 0 \) as the female stock level for which density dependent fertility is equal to density independent fertility. Thus, for a stock level above \( K \), density dependent factors dominate. The compensation parameter \( b > 0 \) indicates to what extent density independent effects compensate for changes in stock size. The equation \((c2')\) implies a recruitment function

\[
    X^0 = r(X^f)X^f = \frac{\hat{r}X^f}{1 + (X^f/K)^b}
\]

which is the so called Sherped-type.

The trophy demand function is specified linear. In addition, it is assumed that the quality effect, as given by the number of males, through the parameter \( \lambda \geq 0 \), shifts the demand uniformly up to:

\[
    q = a e^{\lambda} X^m - \beta h^m X^m
\]
Accordingly, the choke price $\alpha > 0$ gives the maximum willingness to pay with a zero quality effect, $\lambda = 0$, whereas $\beta > 0$ reflects the market price response in a standard manner. The trophy cost function is given linear as well:

$$C = \bar{c} + ch^mX^m$$  \hspace{1cm} (c14')

so that $\bar{c} \geq 0$ is the fixed cost and $c > 0$ is the constant marginal cost. Table 3.9 gives the baseline parameter values.

(\textit{Tab. 3.9 Biological and economic parameter values based on a moose population in Norway})

\begin{center}

\begin{tabular}{lll}
\hline
PARAMETERS & DESCRIPTION & BASELINE VALUE & SOURCE \\
\hline
$\hat{r}$ & max. specific growth rate & 1.15 & Nielsen et al. (2005) \\
$K$ & female stock level where density dependent factors dominate density independent factors & 1000 animals & Nielsen et al. (2005) \\
b & density compensation parameter & 2 & Nielsen et al. (2005) \\
w^0 & average weight young & 60 Kg & SSB (2004) \\
w^f & average weight females & 150 Kg & SSB (2004) \\
w^m & average weight males & 170 Kg & SSB (2004) \\
m^0 & natural mortality young & 0.05 & Nielsen et al. (2005) \\
m & natural mortality female and male & 0.05 & Nielsen et al. (2005) \\
p & meat price & 50NOK/Kg (€ 6.48) & Storaas et al. (2001) \\
$\alpha$ & choke price & 30000 NOK/animal (€ 3885) & Calibrated \\
$\gamma$ & quality parameter demand & 0.001 & Calibrated \\
$\beta$ & slope parameter demand & 60 NOK/animal$^2$ (€ 7.77/animal$^2$) & Calibrated \\
$\bar{c}$ & fixed harvest cost & 500000 NOK (€ 64750.06) & Calibrated \\
c & marginal harvest cost & 2000 NOK/animal (€ 259) & Calibrated \\
\hline
\end{tabular}
\end{center}

For these demand and cost functions we find that the optimal number of hunted males will be

$$h^mX^m = (\alpha - c) / 2\beta$$

without the demand quality effect and when $\mu = 0$ holds the same time (see equation c16).

Finally, table 3.10 reports the results obtained by calibrated data on a moose population in Norway (Skonhoft and Olaussen, 2005).
(Tab. 3.10 Ecological and economic equilibrium for different management regimes, where: $h^0$ is the harvest fraction young, $h^f$ is the harvest fraction female, $h^m$ is the harvest fraction male, $h$ is the total harvest fraction, $X^0$ is the number of young (in 1000 animals), $X^f$ is the number of females (in 1000 animals), $X^m$ is the number of males (in 1000 animals), $X$ is the total stock (in 1000 animals), $\lambda$ is the female shadow price (in 1000 € or NOK per animal), $\mu$ is the male shadow price (in 1000 € or NOK per animal), and $\pi$ is the profit (in 1000 € or NOK per animal).

<table>
<thead>
<tr>
<th>Hunting regimes</th>
<th>$h^0$</th>
<th>$h^f$</th>
<th>$h^m$</th>
<th>$h$</th>
<th>$X^0$</th>
<th>$X^f$</th>
<th>$X^m$</th>
<th>$X$</th>
<th>$\lambda$</th>
<th>$\mu$</th>
<th>$\pi$</th>
</tr>
</thead>
<tbody>
<tr>
<td>No harvest</td>
<td>0</td>
<td>0</td>
<td>0</td>
<td>0</td>
<td>0.33</td>
<td>3.15</td>
<td>3.15</td>
<td>6.63</td>
<td>-</td>
<td>-</td>
<td>-</td>
</tr>
<tr>
<td>Traditional</td>
<td>0</td>
<td>0.26</td>
<td>1</td>
<td>0.29</td>
<td>0.57</td>
<td>0.92</td>
<td>0.27</td>
<td>1.76</td>
<td>0.28€</td>
<td>1.10€</td>
<td>0.53€</td>
</tr>
<tr>
<td>hunting for meat</td>
<td></td>
<td></td>
<td></td>
<td></td>
<td>(2.2NOK)</td>
<td>(8.5NOK)</td>
<td>(4.1NOK)</td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Modern</td>
<td>0</td>
<td>0.28</td>
<td>0.24</td>
<td>0.2</td>
<td>0.57</td>
<td>0.85</td>
<td>0.97</td>
<td>2.39</td>
<td>0.27€</td>
<td>0€</td>
<td>0.59€</td>
</tr>
<tr>
<td>times; Trophy hunt</td>
<td>0</td>
<td>0.25</td>
<td>0.09</td>
<td>0.12</td>
<td>0.57</td>
<td>0.95</td>
<td>2.01</td>
<td>3.53</td>
<td>0.26€</td>
<td>0.25€</td>
<td>0.72€</td>
</tr>
<tr>
<td>No quality effect</td>
<td></td>
<td></td>
<td></td>
<td></td>
<td>(2NOK)</td>
<td>(1.9NOK)</td>
<td>(5.6NOK)</td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Modern</td>
<td>0</td>
<td>0.25</td>
<td>0.09</td>
<td>0.12</td>
<td>0.57</td>
<td>0.95</td>
<td>2.01</td>
<td>3.53</td>
<td>0.26€</td>
<td>0.25€</td>
<td>0.72€</td>
</tr>
<tr>
<td>times; Trophy hunt</td>
<td>0</td>
<td>0.25</td>
<td>0.09</td>
<td>0.12</td>
<td>0.57</td>
<td>0.95</td>
<td>2.01</td>
<td>3.53</td>
<td>0.26€</td>
<td>0.25€</td>
<td>0.72€</td>
</tr>
</tbody>
</table>

As a benchmark, a no-hunting scenario is also included (first row in Table 3.10). Since the young enters the (adult) male and female stages at the same sex-ratio, the number of males and females are here the same. In the traditional regime with meat value maximisation in the Northern Europe, the female harvest rate becomes 0.26, while no harvest of young represents an optimal policy. The marginal harvesting benefit of young is hence the marginal cost in terms of losses from reduced harvesting of males and females. The reason is that the number of females is above the value representing the peak value of the recruitment function and $dX^0 / dX^f = (r'X^f + r) < 0$ without harvesting. The male shadow value is about four times than female shadow value. As demonstrated in the previous section, the male shadow
value is exactly equal its marginal harvesting value $\mu = pw^b$, while the female shadow value is above its marginal harvesting value. However, due to the low female harvesting fraction and an optimal harvesting policy close to the peak of the recruitment function, which is $(r \lambda f + r)$, we have that the female shadow value becomes low.

Table 5.2 shows all the results obtained case by case.

The modern time exploitation scheme is first studied when the quality effect is disregarded; that is $\lambda = 0$ and the inverse demand function (equation c13') reads $q = \alpha - \beta h m X m$. Harvesting down all the males is no longer the optimal policy, and the harvesting fraction is always reduced. As expected, the male-female proportion increases and, at the same time, the female harvesting fraction shifts somewhat up compared to the traditional regime. It is still beneficial, however, to keep the young population unexploited.

When the quality effect is included, the male harvesting rate is further reduced, accompanied by a positive shadow value, indicating that the marginal harvesting income exceeds the marginal cost in optimum. The difference between the male and female shadow values is now quite small. The female harvesting rate decreases somewhat as well. As a consequence, the total stock size is higher when the quality effect is included and, substantially higher than that of the traditional scheme of meat value maximisation. Table 3.10 also demonstrates that the profit increases compared to the traditional regime, and it further increases when the demand quality effect is added. However, for obvious reasons, the specification of the demand function and parameterisation play a critical role here.

Shifting up the meat price $p$, just scales up the shadow price values in the traditional regime. In the trophy hunting regime, with no quality effect and with a zero shadow price value of the males, the harvesting activity and stock sizes will, as explained above, be unaffected as well. On the other hand, with the quality included and $\mu > 0$, the male harvesting activity interacts with the other sex and age stages and hence $p$ has an allocation effect as well. However, as many works have demonstrated (see, for example, Storaas et al., 2001), sensitivity analyses show that the female harvest rate increases only modestly even for a quite dramatic price increase. The reason is that female stock is close to the peak of the recruitment function.
3.4.2 Conclusions

Finally, in this last third approach we analysed a three stage model of the Scandinavian moose with density dependent fertility. The two exploitation schemes have been studied and it is demonstrated that harvesting down the whole biological “end” product, i.e., the (adult) male population in this model, always represents the best option when meat value maximization is the goal. In the numerical examples, this option is accompanied by zero harvesting of the young and modest female harvesting. Within this regime, the biological notion of females as “valuable” and males as “non-valuable” is easily recognized even if the shadow value of the males might be higher than that of females. The modern times exploitation scheme with a market for trophy hunting, changes the optimal harvesting condition of males. Hunting down the whole population will no longer be the best option suggested that a well developed market for trophy hunting is present. In addition, the trophy hunting market allocation also spills over to the conditions for meat value maximization of young and females. The male-female population ratio will increase, and more female harvesting may take place.

Although the model presented in this third approach is enough simple, it encompasses some general results that will survive in more complex stage structure models. Most important, we have highlighted the economic forces influencing harvest in three different stages that, in various degrees, are present in many structure population models. In the model we showed for a Scandinavian moose population there are two recruiting stages that recruit in different ways. The young represents a value through recruitment to the (adult) male and female stages. As long as density dependent growth factors are weak, or non-existing (as in this case), harvesting young does not pay off. For the females, on the other hand, a traditional trade-off between recruitment and harvest is present through the density dependent fertility mechanism. This mechanism will also be showed in more complex models. Finally, the (adult) male stage is considered as the biological “end” product, and thus does not influence recruitment. It is therefore tacitly assumed that there are always enough males for reproduction.
3.5.1 Modelling a fallow deer population structure and harvesting by Vensim Software DSS: a simulation approach

Vensim DSS is a visual software to help conceptualize, build, and test system dynamics models (Ford, 1999). It was created by Ventana Systems Inc. (149 Waverly Street, Belmont, MA 02178) as a “high-end tool” to support the company’s consulting projects for governments and business:

«In order to do this work well, we needed tools that would support the efficient development of high quality models. Because we were not able to find these tools, we developed our own. Vensim was born as a high-end tool to speed the thorough analysis of models (Ventana, 1996)».

Vensim’s collection of high-end features is impressive. They include optimization, Kalman filtering, macro programming, flight simulators, and risk analysis.

We tried to use Vensim to simulate the behaviour of our fallow deer population under a natural growth rate, in absence of natural predators (i.e wolves), and under annual culling by hunters or Park rangers.

The history of our fallow deer population was yet described in the first part of this chapter.

The best way to explain the nature of the “dynamic problem” is to draw a reference mode (see Fig. 3.4).

The starting step is to construct a stock-and-flow diagram to show a minimum combination of stocks and flows that can explain the reference mode. Our steady point is the fallow deer population, and the changing variable is the size of the culling. The number of fallow deers is increased by births (depending by a growth rate and constructed by the carrying capacity of the place), and decreased by the harvesting plan (depending on catchability coefficient and effort employed). In our growth rate function we consider a mortality rate to simulate a more realistic condition in nature.

We implemented the first model with a second one considering the profit problem: the net profit will be a combination of revenues, costs, discount factor, and time variable expressed as a shadow variable.

The combination of the two models, the nomenclature and equation used are shown Tab. 3.11.
By Vensim Software we tried to manipulate our fallow deer population in all the ways and conditions we will, combining both the population growth as the size of the cull, with the desired profit. Anyway, as the possibility by Vensim to produce tables and figures is enormous, it is unrealistic to show some of the possible simulations in a paper presentation (optical supports could be more useful in this case).

Softwares like these are still few known in Italy; but their potential work could growth up in the future of the wildlife techniques, and not only. People approaching to Venisim Software have to start from a number of written hypotheses and developed a model that has helped us to explore some of these hypotheses in a unified framework. By continually relating a model to hypotheses, this issue is addressed from the beginning and can be less daunting in the end.

(Fig. 3.4 Vensim Software simulation model)
(Tab. 3.11 Nomenclature used in Vensim simulation model and following equations)

- **Fallow deers** ($X_t$): initial population density
- **Net growth function**: $F(X_t) = rX_t(1-X_t/K)$
- **Growth rate**: $r$ (*constant*)
- **Carrying capacity**: $K$ (*constant*)
- **Harvest**: $Y = qX_tE_t$
- **Catchability coeff.**: $q$ (*constant*)
- **Effort**: $E = $ wage/hours
- **Annual Profit**: Revenue-Cost ($\text{€/Year}$)
- **Net Profit**: Annual Profit* Discount ($\text{€}$)
- **Discount**: $\rho \times \text{time}$ (*constant*)
- **Discount rate**: $\delta$ (*constant*)
- **Cost**: Marginal Cost*Effort
- **Marginal Cost**: $\text{€/Year}$ (*constant*)
- **Revenues**: Harvest ($Y = qX_tE_t$) ($\text{€/Year}$)
- **Price**: $\text{€/Kg of venison}$ (*constant*)

4. Discussion

Bioeconomic analysis of wildlife resource/pest management is based on models that predict how pest density and the resource/pest affect respond to different strategies for the managing control. Various analytical techniques can be applied to these models in order to draw useful information about resource/pest management from them. Anyway, the usefulness of this information will be limited by the realism of the underlying bioeconomic model.

In our modern times, the meaning of this consideration lies in the correct biological and economic analysis of public decisions about wildlife management, which should have to include three main aspects: 1) the value to users of increases or decreases in wildlife populations; 2) the relationship between the current wildlife stocks and the future wildlife populations; and 3) the costs of providing increments of wildlife populations through habitat manipulations and/or management alternatives.
All these considerations are generally found in a “bioeconomic” approach to the analysis.

The main case study presented in this work aimed to provide a bioeconomic model for a fallow deer population, considered both a value and a pest, in a Natural Park in Italy. Deers, as many other wild species compete with humans for the land use and other nutritional resources. From always in the history these were the main reasons of conflicts of interests between humans and wild animals. If certain qualities of a deer herd are to be maintained, humans must help control deer populations. When nutrient limitations alone control deer populations, most herd health qualities suffer. Historically, predation was probably the primary influence. Thus the importance of man’s role as a deer predator has increased. The only relatively efficient means to manage deer herd numbers other than habitat modification is through managing hunter harvest (i.e., contemporary predation). Management of deer numbers is necessary to regulate or control deer herd health.

Deer harvests on an area are generally managed through harvest quotas. Deer harvest quotas are the numbers of male and female deer that can or should be removed from an area during a certain time frame. When establishing harvest quotas, managers should consider goals, habitat limits, deer density, sex ratio, fawn crop, and recent trends in deer population parameters. Even if all deer population parameters are the same on different management areas, deer harvest quotas generally vary when the goals vary. Probably solutions are represented by the following goals, expressed in few words: potential management goals, meaning that there is more than one way to manage deer; eliminating deer or minimizing deer numbers; sustaining a viable deer population at relatively low densities; seeing lots of deer; optimizing deer herd health and numbers; sustaining maximum harvest of deer (or bucks); sustaining maximum harvest and quality of trophy bucks.

All the models presented in this works demonstrate that wild animal harvest policy depends critically on economic conditions of the area. That’s the reason as it is desirable that future works on wildlife management will try to manage the problem in an ecological, economical and not finally, social perspective.
References


