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

PhD student: Irene Di Vittorio
University of Verona, Italy
Supervisor: Prof. Federico Perali, University of Verona, Italy
Tutor: Dr Piran White, University of York, U.K.
Co-Tutor: Dr Jim Smart, University of York, U.K.
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 economic 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 bioeconomic 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. At the end a sensitive analysis will be carried out to check which variable determines a “sensitive” change in the wild population structure and in the economic outputs for the protected natural area. 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.
**Introduction**

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.

But if we consider that many wildlife populations could cause damage to agricultural and forestry systems, on the other hand, depending on the ecological-economic condition, they can also be 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 a broader context, it is the damage of vertebrate pests that justifies their economic control.

The management of undesirable pest species involves making choices that will determine the costs of measures to limit the abundance or distribution of the species, and the benefits they will deliver. To make this choice defensible, the effect of alternate courses of action on the costs and benefits of management should be understood. And to make evidence on how costs and benefits vary between alternate management strategies, the biological and management components of a pest/resource system must be linked in such a way, that its economic inputs (costs) and outputs (benefits) can be estimated and compared.

Clark (1973, 1976, 1990) coined the term “bioeconomics” to describe the economic analysis of managed biological systems. In other words, he described a mathematical framework that linked the dynamics of managed biological resources, with the economic imperatives driving management. Clark’s analysis has been extended on field applications, and dynamic programming, simulation, and optimization models are now used to achieve the optimal balance (Choquenot and Hone, 2002). The use of simulation models is particularly useful where the behaviour of the managed system is stochastic or uncertain, and this uncertainty constitutes a significant economic risk (Williams, 1989; Hilborn and Mangel, 1997).

Regardless of how they are actually implemented, the principles of bioeconomic provide a powerful approach to the analysis of management options for biological systems. As such, it is surprising how few examples of their application to invasive/resource species management have been reported (Hone, 1994; Choquenot and Hone, 2002). Moreover, it results more surprisingly
that in Italy there are no information, in past and current works, about this kind of analysis applied on wildlife management.

In this paper, we will briefly review the aspects of bioeconomic analysis relevant to pest/resource species in a natural Park, where a fallow deer population is subject to harvest every year to achieve an approximate balance between the maximum sustainable yield (to maintain the population in the Park at levels sustainable by the carrying capacity) and the costs, sustained by the annual Park policy plans, provided by deer damage to forestry and the following costs of control (including the costs of harvesting traps, costs of rifles, cartridges and gasoline for the Ranger’s cars, costs of harvesting effort, indirect costs deriving by damage to woodland and crops inside the Park, costs of assurance deriving by deer-vehicle collisions).

**Bioeconomic analysis**

When deer significantly damage crops, forests, or vehicles, they are considered to be a nuisance (Alexander et al., 2003; Dawson, 1970). The best approach to control deer damage is an integrated pest management (IPM) plan, which includes careful monitoring of any one, or combination of, the following strategies: population management, fencing, repellents, or vegetation management (Moore et al., 1999). Although non-lethal techniques can help to minimize damages caused by deers on the environment, the lacks of any population control will likely result in an increasing population, and the problems associated with this increase.

Controlling deer damage requires a comprehensive program. If from one hand deer can have a substantial impact on woodland vegetation, on the other hand they play a significant role in woodland ecosystem function. In the absence of a selective control, deer populations can rise to very high densities due to lack of predators and to the regular provision of ideal habitats through felling and planting in woodlands. Management of deer populations is necessary to limit their impact, particularly in ancient semi-natural woodlands. When deers have an impact on biodiversity (in general) of such a way, selective pest control strategies become a necessary tool in management plans. In these conditions, bioeconomic models applied on pest management make the point on a range of approaches available for assessing the economic performance of alternative pest management.

The basic construct for a bioeconomic analysis applied to invasive/resource species management is outlined in *Fig 1*. Economic inputs (costs) to the management system are generally dominated by the cost of controlling the pest species, to limit its abundance or distribution. Economic outputs (benefits) from the management system accrue through
additional income, where the invasive species affects resources of realised or potential income, or the improved condition of public assets, where these do not have a readily realised marked value. Regardless of whether the invasive species affects income or asset condition, the costs and benefits from its control are brought together in some form of benefit/cost analysis.

(Fig. 1 Conceptual diagrams on how bioeconomic analysis of invasive species management would generally be structured. A model, which links pest control, pest abundance, and the benefits associated with changes in pest abundance, is used to estimate the value of economic inputs (costs) and outputs (benefits), which accrue from pest control activities, which are contrasted in some form of cost/benefit analysis.)

The biological model: an age-structured population model

Introduction

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.
Only a small number of studies have been attempted to optimize harvest when the real initial demographic structure of the population is markedly different from the equilibrium structure. In such cases, transient dynamics are of particular importance (Fox and Gurevitch, 2000). When sequential harvest decisions are made over time, transient dynamics may have a large influence on observed population change between decisions, and so must be accounted for during the decision-making process. Harley and Manson (1981) were the first to discuss how a structured population might be most efficiently brought to equilibrium in a finite number of steps. A small number of other studies (Stocker, 1983; Milner-Gulland, 1997; Hauser et al., 2005) have used stochastic dynamic programming to determine the optimal harvest decision for any possible initial population structure and abundance. However, these studies have divided the target population into only a small number of classes, and have generally assumed complete knowledge of the structure of the population at the time of harvest. While modelling a small number of classes may have been adequate for these studies, incorporating more classes is likely to be limited by data constraints and the computational method.

When we attempt to manage a natural renewable resource, what is important to keep in mind is that the exploitation of a part of the resource (causing damage, or spreading disease, etc..) have to be a sustainable yield for the 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 a balance population dynamics and the other ecological and economic components taking in act; which are the cost of harvesting by land owner or by the policy of a Natural Park; which are the revenues derived by the exploitation of the 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. This depends on the biological characteristics of the species, the costs of harvesting, and the price received for the products of harvesting (venison, trophy). Thus the decision-making of harvesters is usually expressed in monetary terms; the obvious reason is that they are aiming to maximize their profit. Other influences on decision-making can also be included in the model, such as the user’s attitude to uncertainty or their wish to retain the
population above a certain size for conservation reason. The work here presented to analyze the case of a reserve into a Natural Park in Italy, in which the profit from harvesting and selling wildlife population has to equal the revenues by it, under the limits of the current Italian policy for wildlife populations in a National, Regional and District Park, which considered them as property of nobody.

**Biological population dynamics model**

Population modelling is a tool highly used by wildlife manager. In mathematical symbolism, a population model can be expressed as

$$N_{t+1} = N_t + B_t - D_t + I_t - E_t$$

It means that population size ($N$) at time $t+1$ is equal to the population size at time $t$ plus birth ($B$) minus deaths ($D$) plus immigrants ($I$) minus emigrants ($E$).

It could be useful to give briefly some information on the developing of a biological model for annual species. In this case all adults alive in year $t$ die before year $t+1$. Thus in the following equation

$$N(t+1) = f N(t)$$

we express birth as fecundity rate, $f$. This rate may be thought as the average number of individuals born per individual at time $t$ that survive to be counted at the next time step $t+1$.

For annual species, fecundity is equal to the growth rate of the population,

$$N(t+1) = N(t) R$$

The growth rate of the population is often used to predict the increase of the population size in the future, and this equation could be expressed by

$$N(t) = N(0) R^t$$

It means that if we want to predict the population size for two years, we have

$$N(t+2) = N(t) R^2$$

This equation represents a model for the dynamic of populations. A simple model such this one is an equation describing the relationship between independent variables, parameters, and dependent variables. The dependent variable (or state variable) is the quantity we want to estimate (such as the future population size). Obviously, it depends on other factors, called independent variables. The populations size at time $t$, $N(t)$, is the dependent variable, and time ($t$) is an independent variable. The growth rate ($R$) and the initial population size, $N(0)$, are parameters. The type of population growth described by this model is called exponential growth model. Sometimes it is also called geometric growth or Malthusian growth.

Anyway, the most of mammal species are not annual; they survive for more than one year and reproduce for more than once. To allow for the survival of individuals for more than a
single time step, we may introduce a survival rate \((s)\), which is the proportion of individuals alive at time \(t\) that survive at time \(t+1\). Thus, the population size at the next time step is the sum of two numbers: (1) the number of individuals that survive to the next time step (out of those that were already in the population), and (2) the number of offspring produced by them that survive to the next time step. It could be written in this way:

\[
N(t+1) = N(t) \ s + N(t) \ f
\]

This is a model of population growth in which births and deaths are expressed as fecundities and survivals. By rearranging the formula, we observe that

\[
N(t+1) = N(t) \ (s + f)
\]

And so \(N(t+1) = N(t) \ R\)

In other words, the last expression explain that

\[
N(t+1) / N(t) = R
\]

While the growth rate is given by the last equation, the percentage of variation of the growth rate is given by

\[
[(\text{Final Pop. – Initial Pop}) / \text{Initial Pop.}] \ *100.
\]

In the same way as in the upper equation, we predict how long will the average rate of increase take for the population to double in size:

\[
R^t = 2
\]

\[
t \ ln \ (R) = ln \ (2)
\]

\[
t = ln \ (2) / ln \ (R)
\]

Someone may use calculations of the growth rate of the population to estimate the number that may be removed in perpetuity. The basic idea is that if things continue in the future as they have in the past, one could remove or harvest a number of animals from a population so that the effective growth rate is 1. Whenever a model is constructed, it employs a set of assumptions reducing the complexity of the real world to manageable proportions.

The following assumptions should have to be consider: 1) the model for exponential population growth is a deterministic model; 2) the population abundance can be describe by a real number; 3) population grows or declines exponentially for an indefinite period; 4) births and deaths are independent of the ages or any other unique properties of the individuals; 5) the model is also called a “birth-pulse” model as we assume that all the offsprings are born in the same period (May – June).

To have a clear knowledge about population growth model we have also to consider the case in which populations reproduce and die continuously, as does in human population. The basic model for population growth in discrete time is
\[ N(t + 1) = N(t) + B - D \]

This could be rewritten as
\[ \Delta N = N(t+1) - n(t) = B - D \]

The symbol \( \Delta N \) is the difference in population size. If the time interval represented by \( \Delta N \) is small, we can approximate it by the derivative \( dN/dt \).

In the other case, rather than express births and deaths as numbers of individuals, they may be expressed as instantaneous rate, giving
\[ dN/dt = bN - dN \]
\[ = (b-d)N \]
\[ = rN \]

The difference between the birth rate and the death rate in a continuous time is called instantaneous growth rate \((r)\). The equation above may be solved in this way
\[ N(t) = N(0) e^{rt} \]

By the analogy with the equivalent discrete time equation, we can observe that
\[ R = e^r \]

We can now consider species for which the age of an individual is one of the most important characteristics having strong effect on the individual’s chances of survival and reproduction. Chances of successful breeding are also dependent on age in many species. Models like this in which we are going to consider different age and sex classes are called age-structured models. It could be important to precise that “age” and “age-class” are different concepts, although they are both consider as variables; but the first one is a continuous variable, while the second one is a variable in a discrete time model.

In these new kind of models there are two restrictions to consider. First, the time step of the model and the interval of an age class must be the same. The second restriction is represented by the fact that all age classes must have the same with or interval.

Before going to develop an age-structured model we should have to consider some assumptions regarding it. From a practical point of view, using an age-structured model implies that one can determine the age of all individuals in the population with certainty. The basic assumption of age-structured models is that the demographic characteristic of individuals are related to their age, and among individuals of the same age, there is little variation with respect to their demographic characteristics, such as chance of surviving, chance of reproducing, and number of offspring they produce. Initially, we also assume that the population is closed, i. e. there is no immigration or emigration.
Simple age-structured population models make some additional assumptions: there is no demographic or environmental stochasticity; there is no density dependence. We will assume that reproductions take place at the same time. Such behaviour is a typical anti-predatory strategy used by many animals, and such a model is called “birth-pulse” population model. It could also be possible to develop age-structure models with the alternative assumption that births occur continuously (as in humans, for example); this kind of models are called “birth-flow” models.

The general form of survival rate \((S_x)\) between each age-classes is given by the equation

\[
S_x(t) = \frac{N_{t+1}(t+1)}{N_x(t)}
\]

In other terms, the number of individuals that survive to be, for example, one-year-old \((N_1)\) is the number we counted as zero-year olds \((N_0)\) times the survival rate of zero-year old \((S_0)\).

The most common mathematical tool to construct a biological model (particularly if there are lacks in the data, and we cannot use a life table which requires more details) is the “Leslie Matrix” (Leslie, 1940). 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 are 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.

At this point, multiplying the age distribution by the Leslie Matrix is the same as multiplying it by a scalar number. Mathematically, this is

\[ N(t + 1) = L \cdot N(t) = \lambda \cdot N(t) \]

where \( \lambda \) is a special number called the dominant eingenvalue of the matrix. Once at its stable age distribution, the population grows exponentially with the rate \( \lambda \). If the logarithm of the population size is plotted against time, the slope of the line, after the stable population structure has been reached, is equivalent to \( \ln \lambda \), the intrinsic rate of natural increase. Briefly, it means that this eingenvalue indicates that the population is capable of increasing, and the natural logarithm of the eingenvalue is an estimate of the intrinsic rate of increase:

\[ r = \ln \lambda. \]

The dominant eingenvalue can also be used to estimate the number of individuals that could be removed from the population, to bring it back to the initial population size, by the equation:

\[ H = 100 \left( \frac{\lambda - 1}{\lambda} \right) \% , \text{ where } H \text{ is expressed as percentage of the total population.} \]

Note that this equation cannot be used to estimate the number of animals that should be removed from a population to reach the aim of the management plan.

At this point we know that the basic assumption of age-structured models is that the demographic characteristic of individuals (such as fertility and survival chances) are related to their age, and among individuals of the same age, there is little variation with respect to these demographic characteristics.

In an age-structured model, there are only two types of transitions: an individual may get older (i.e., move to next class), and/or it may produce offspring (i.e., contribute to the first age class). If neither of these happens, the model assumes that the individual died. The two types of nonzero elements of the Leslie Matrix represent these two types of transitions: those at the sub-diagonal representing the survival, and those in the first row, representing fecundities. Moreover, we assume that \( F_0 = 0 \), which means that the youngest individuals do not reproduce (which is often the case). We can describe this age-structured model with a diagram (Fig. 2), in
which boxes represent age classes and arrows represent transitions (survivals and fecundities) from one age class to another.

So, given the Leslie Matrix as below

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

the corresponding diagram is

(Fig. 2 Diagram of an age-structured model)

When we combine three-year-old and older individuals into a composite class, their survival rate was represented by the matrix element at the lower-right corner of the matrix below. We will use the symbol \( S_3^+ \) for the survival of three-plus-year-old individuals. When these individuals survive for another year, they are still counted in the same class. In Fig. 3, it is represented by the loop around the box for “3+ year old” individuals.

The survival rates of the 3+ age class from one year to the next is given by

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

And the corresponding matrix is
\[
L = \begin{pmatrix}
F_0 & F_1 & F_2 & F_3 \\
S_0 & 0 & 0 & 0 \\
0 & S_1 & 0 & 0 \\
0 & 0 & S_2 & S_3 +
\end{pmatrix}
\]

(Fig. 3 Diagram of an age-structure model with a composite age class for individuals three-years old and older)

**Costs of control**

The cost of invasive species control is the product of how many individuals should have to be removed from a population to limits its abundance or distribution, and the cost associated with each of these removals. Techniques that can plausibly be applied to invasive pest species fall into three broad categories: (1) direct removals, where the control measure targets individuals; (2) indirect removals, where measures are placed into the environment to intercept and remove organisms (traps, poison, etc.); indirect system modifications, where measures modify the environment or pest population in such a way that the efficiency of other control measures is enhanced (barriers to dispersal, release of sterile individuals, etc.).

Over-and-above a fixed capital investment in materials, costs associated with direct removals are dominated by the variable components of time spent accessing areas to which they will be applied and then undertaking the activity. Once an area is accessed, the cost of removing pests is generally a function of their prevailing abundance, more time being required to find each pest to be removed as pest density declines (Choquenot et al., 1999). This produces a curvilinear relationship between pest abundance \( (N) \) and the cost per removal \( (C_k) \), which represents the fixed costs of handling each pest at high pest densities and at low pest densities (Fig. 4).
Choquenot et al. gave a useful formulation in describing this type of relationship (1999):

\[ k = c + e^{-(N-b)i} \]

where \( c \) are the fixed costs associated with each removal (equivalent to the costs that accrue for each removal when the rate of removal is maximized), \( b \) is the abundance of pests below which no pests can be theoretically located for removal, and \( i \) summarizes the effect that declining pest abundance has on the cost of each removal. This formulation is based on a standard functional response model that relates the rate at which prey are consumed by a predator to prey abundance (Hone, 1994).

The number of pest animals that have to be removed from a population to limit its distribution or abundance to acceptable levels, is determined by: (1) the number of individuals in the population; (2) the history of control activities applied to the population; (3) the resilience of the population to control measures (i.e., its rate of recovery). There is a range of models that are used to understand and predict how populations might respond to impose reductions in their density. Very common set of models that have been broadly applied to fish populations are the logistic-based models (Hilborn and Walters, 1992).

These models are now highly applied on the management of terrestrial species as well as on fishery (Milner-Gulland, 1997; Milner-Gulland and Mace, 1998).

The dynamics of wild animal populations that conform to the assumptions of logistic growth can be summarized through the instantaneous or lagged effect that prevailing population abundance (\( X_t \)) has on its exponential rate of growth (\( r \)).
The basic equation of a logistic growth model is given by

\[
\frac{dx}{dt} = rx(1 - \frac{x}{K}) = F(x) \quad \text{(Eq. 1)}
\]

The general solution to this equation is represented by the following one (where \(x_0\) is the population at time zero, also called the initial population):

\[
X(t) = \frac{Kx_0e^{rt}}{K + x_0(e^{rt} - 1)}
\]

We can now suppose that the population described by the logistic equation is subject to harvesting at a rate \(h(t)\), and we get

\[
\frac{dx}{dt} = F(x) - h(t), \quad 0 \leq h(t) \leq f(N(t)) \quad \text{(Eq. 2)}
\]

In many cases we can find this equation written as \(N(t + 1) = f(N(t)) - h(t), \ 0 \leq h(t) \leq f(N(t))\).

Some important considerations could now be made: 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 the 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 \(h = 0\).

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 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 a constant and is called the catchability coefficient.

Considering now the Cobb-Douglas production function, defined as the rate of output of a certain productive process in terms of inputs to the process (Chiang and Wainwright, 2005), and expressed in the form \(Y_t = q X_t^\alpha E_t^\beta\) where the coefficient \(\alpha = \beta = 1\), we can substitute Eq. (3) into our basic harvesting model, Eq. (2), we obtain a new equation

\[
\frac{dx}{dt} = F(x) - qEx = rx \left(1 - \frac{x}{K}\right) - qEx. \quad \text{(Eq. 4)}
\]
With the price of animals represented by $p$, $q$ is the catchability coefficient, $c$ are the harvesting unitary costs, the private property equilibrium (PPE) is the equilibrium solution $(X_{PPE}, h_{PPE}) = (X_{PPE}, SY(X_{PPE}))$ which maximizes the rent as follows:

$$\max_{X \geq 0, h = SY(X)} \left[ ph - \frac{cqh}{X} \right]$$

The situation is different in the common property equilibrium $X_{CPE}$, which makes the rent null, and is given by

$$X_{CPE} = \frac{cq}{p}.$$  

The last condition is typical of the Italian Parks where wildlife animals are considered, by the current Italian policy, property of the State and of nobody, and the management plan of the Park cannot accrue revenues by the animal harvesting and selling higher than the costs to manage them.

In other terms, it means that

$$\frac{dx}{dt} = F(x) - qEx = rx \left(1 - \frac{x}{K}\right) - qEx = 0,$$

$$TR - TC = pqEx - Ec = 0.$$  

These equations can be readily solved for the equilibrium-effort level $E = E_{\star}$:

$$E_{\star} = \frac{r}{q} \left(1 - \frac{c}{pqK}\right),$$

and for the corresponding stock level $x = x_{\star}$:

$$x_{\star} = \frac{c}{pq}.$$  

In case study analysis presented in this work we will explore in more details a bioeconomic model in which the current profit will be described by

$$\pi_t = pH^t - c E^t,$$  

where $E$ is the effort used and $q$ is the catchability coefficient.

Considering a modified Cobb-Douglas function, which gives a description of the culling process for deer populations, we could obtain the following equation:

$$K = q \cdot E^\alpha \cdot N^\theta \cdot L^\gamma$$

Here $K$ is the size of the cull (and not the carrying capacity denoting by $k$), $q$ is a scaling constant, $E$ is the effort applied (number of guards or stalkers per year), $N$ is the total population size, and $L$ is the land area over which the herd is dispersed (in unit of acres, for compatibility with Beddington and Taylor’s work, 1973). Exponent $\alpha$ is the effort elasticity, and it is assumed
to be $0 < \alpha < 1$; $\beta$ is the herd size elasticity and it is assumed to be $0 < \beta < 1$, and $\gamma$ is the land area elasticity, and we usually assume that $\gamma = 1$.

When we assume that harvesting takes place instantaneously, and assuming a generalized Schafer function, we have.

$$H_t = q E^{\alpha} [X_t + F(X_t)]^{\beta}$$

This inserted into the current profit expression above gives

$$\pi_t = p H_t - c \{H_t / q [X_t + F(X_t)]^{\beta} \}^{1/\alpha}$$

Maximization to respect to $H_t$ for given stock size yields the optimal harvest (in number or biomass of individuals) as

$$H^*_t = q^{1/(1-\alpha)} (p\alpha/c) \alpha^{a/(1-a)} [X_t + F(X_t)]^{\beta/(1-a)}$$

We can now find the optimal harvest fraction as

$$h^*_t = q^{1/(1-\alpha)} (p\alpha/c) \alpha^{a/(1-a)} [X_t + F(X_t)]^{(\alpha + \beta - 1)/(1-a)}$$

Note here that $\alpha = \beta = 0.5$, $H^*$ is linear in stock, and that $h^*$ is fixed over time for $\alpha + \beta = 1$.

Next, when plugging into the population growth equations, we find the stock growth as

$$X_{t+1} = X_t + F(X_t) - [X_t + F(X_t)]h^*$$

Now we can assume that in a place where the resource in question results in zero profits for our managers (in an Italian Park, for example, as well as in for an open access to the resource) our harvesting results in

$$H_{t}^{CPE} = \left(\frac{qp^\alpha}{c^\alpha}\right)^{1-\alpha} (X_t + F(X_t))^{\beta} 1/1-\alpha$$

So now we have the harvesting fraction as follows

$$h_{t}^{CPE} = \left(\frac{qp^\alpha}{c^\alpha}\right)^{1-\alpha} (X_t + F(X_t))^{\alpha + \beta - 1} 1/1-\alpha$$

Note again that for $\alpha = \beta = 0.5$, $H^*$ is linear in stock, and that $h^{OA}$ is fixed over time for $\alpha + \beta = 1$.

In steady demands, as well as, we have to manage in an Italian Park under the current environment policy, the profit function is equal to zero, in the way in which

$$\pi = pH - c(H / q X^\beta) \{1/\alpha = 0$$

where $X$ is total available stock, as described above. This gives the steady state equilibrium open access harvest (equivalently in Italian Parks):

$$H_{CPE}^{CPE} = \left(\frac{qp^\alpha X^\beta}{c^\alpha}\right)^{1-\alpha}$$
Moreover, we can note that $H^* = \alpha \frac{\alpha}{\alpha - 1} H^{OJ}$ is well expressed by Fig. 5. Considering ISYF as the Induced Sustainable Yield Function, which gives the ecological equilibrium relationship between the resource abundance in the reserve and the harvesting, we have:

(Fig. 5)

**Case study**

**Study area and population**

The present work aims to analyze the ecology of a population of fallow deers in a reserve into a regional Park in Italy which, in absence of natural predators and without the possibilities to dispersal outside the Park caused by fences and other artificial barriers, cause 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 and the damage caused by the same population. As the fallow deer population live within a Park (the situation will be different if we have to manage a wild population in a private property), the management plan is constrained by the current Italian policy, which not provides profit by the wildlife management. This reserve is an old hunting Estate, holiday house of Kings and Presidents; it became part of this Regional Park only in 2000.

The study was carried out in the San Rossore Estate in central Italy (43°43’N; 10°19’E). The 4.653 ha study area lies on the Tyrrhenian Sea coast, west of Pisa, Italy. Bounded by the Serchio river to the North, the Arno river to the South, the Tyrrhenian Sea to the West, and fenced in on the East, it is characterized by a sub-Mediterranean climate, with mild winters and warm and dry summers, and a plain orography. The ground is mainly sandy, and the area available to deers
consists of grassland, open or bushy, deciduous and mixed woods (*Quercus* spp., *Fraxinus* spp., *Populus alba*, *Populus canescens*, *Almus glutinosa*), pine woods (*Pinus pinea* and *Pinus pinaster*) and marshes; the remainder is sandy littoral vegetation. Moreover, 1.033 ha of fenced agricultural areas are not available to deers.

Ten different types of vegetational habitats are present in San Rossore Estate (Ciuti, et al., 2004). The number of fragments for all typologies (137) supports the observation that there is a non-uniform distribution of resources in this environment (*Fig. 6*).
((Fig.6 Study area map shows sectors considered in fallow deer sexual segregation analyses. The small maps above from the left, show the location of fenced areas, different types of vegetational habitats present in San Rossore (pine woods PW, deciduous and mixed woods DMW, meadows ME, mixed pine and deciduous woods PDW, maritime pine woods MPW, marshes MA, degraded coastal zone DCZ, beach B, oak plantations OP, poplar plantations PP, and road networks. The small table below shows the availability of different types of vegetational habitats included in the western undisturbed sector and in the eastern disturbed one. The square marked called as lek indicates the 4 ha central area of the rutting site))

Some typologies are more represented, especially mixed deciduous forest and domestic pine, two habitats fragmented throughout the study area, than are others. Results by Bruno and Apollonio, 1991 explain only five habitat types contain resources usually selected by deer (deciduous and mixed woods, DMW, mixed pine and deciduous woods, PDW, meadows, ME, oak plantations, OP, and marshes, MA). The habitat type DMW proved to be subdivided on 43 fragments (average fragments area 23.9 ha). The habitat type ME, selected by deer during autumn and spring (Bruno and Apollonio, 1991; Apollonio et al., 1998), proved to be only 11.55% of the total available area to deer and was subdivided on 10 fragments (average fragments area 41.9 ha). The habitat type OP is an area with high trophic resources availability, but it represents only 1.29% of the available area to deers. The other habitat types (pine woods, PW, maritime pine woods, MPW, degraded coastal zone, DCZ, poplar plantations, PP, and
beach, B) proved to be lacking in trophic resources used by deer (Bruno and Apollonio, 1991): it is to be noted that habitat type PW is interspersed with other types rich in food availability, increasing therefore the degree of fragmentation.

During the week, the estate is closed to the public, while designated areas are open at weekends. However, also during the week, the eastern side of the estate is characterized by intense anthropic disturbance caused by the presence of farmers, workmen and horse riding and, recently, by tourists having access to a visitors’ building: the two entrances to the estate, the two inhabited areas, and the main road are localized in this part of the study area.

The eastern side of the estate, a 549.5 ha of fenced area, was subdivided into 12 zones (averaging 45.4 ± 2.7 ha) named zone 0–100 m, zone 100–200 m, etc, and zone 1,100–1,200 m. Each zone includes parts of the estate that are the same distance from the fence (for example, zone 500–600 m includes places that are between 500 and 600 m from the fence). The whole area was called zone 0–1,200 m (Eastern sector, Fig. 6). The remainder of the Estate (zone over 1,200 m, Western sector) was definitely less affected by human activity. It is to be noted that the disturbed area contains almost only habitat types selected by deers (DMW, ME, OP) subdivided on few fragments. This area contains seven of ten fragments of the habitat type meadows, and five large fragments of the habitat type deciduous and mixed woods. The disturbed area proved therefore to be a more homogeneous and less fragmented environment (Shannon index value = 1.99) (Krebs, 1989) than the other areas of the estate (Shannon index value excluding disturbed area = 2.68). A thorough analysis of Eastern-sector habitat composition revealed that we can consider two different sub-zones following the distribution of habitat types: the zone 0–500 (68.3% DMW, 19.9% OP, 10.9% ME, 0.8% PW) and the zone 600–1,200 (39.0% ME, 32.3% PW, 21.2% DMW, 6.8% PDW, 0.7% MA), which were subdivided by an intermediate sash mainly composed of meadows (zone 500–600: 85.7% ME, 3.2% PW, 10.9% DMW). Large predators are absent and the wild boar (Sus scrofa) is the only other ungulate. Fallow deer are shot by game keepers both on foot and from the car, with a prevalence of the second method, using all small secondary roads that are homogeneously distributed throughout the whole estate (Fig. 6).

Moreover, autumn and winter captures are used extensively following yearly management plans. These activities are conducted everywhere on the estate using an unique technique of harvesting alive animals.

In San Rossore, the fallow-deer population adopt a mixed mating system: some males defend a territory on a lek, others a single, isolated territory (Apollonio et al., 1998).
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 four times at dawn and early morning. Deer density within study area was established to be about 26 deer/100 ha.

**Methods and results in progress**

Ecological and economic data were collected from 2001 until 2004. 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 *yearling*; 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*.

The following table shows the times during which census and removing control programs are carried out, as well as new offspring are born.

<table>
<thead>
<tr>
<th>G</th>
<th>F</th>
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<th>J</th>
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*Censuses are carried out during the last days of March and the first ten of April, when bucks (the adult males) have lost the antler and it is growing up, as every year this process takes place. Births usually occur during May and June, although they can continue up to September, and only one fawn is born. Fallow fawns, at birth, weight approximately 4.5 kg, and they can be as varied in colour as the adults but all will be spotted to a greater or lesser degree. Adult males, or bucks, stand about 90-95 cm at the shoulder and can weigh upward of 70 kg, depending upon the time of the year and health condition. Does (adult females) are slightly smaller than bucks, standing 80-85 cm at the shoulder and weighing about 45 kg. The breeding season, depending on weather seasonal conditions, usually occurs from September to November.*
We usually use two strategic harvesting plans to control the number of fallow deer in the Estate: the first one consists in the use of “fenced capture sites”, by which animals alive are removed. These animals will be sold to landowner of private hunting estate at higher prices than culled animals. The second technique is a shooting control by the Park Rangers. Both of these two methods have advantages and disadvantages, in costs and in difficulty to apply them. The harvesting plan takes place from the end of October until February-March, following the hunting timetable; anyway, when particularly high damages to crops occur, deer can be shot during all the year.

During the four years of fieldwork we collected data on the number of animals counted in April; the number of shot and harvested alive animals. The price of the animal is considered in terms of Kg/Euro, but previously the animal weight price is fixed in a public auction sale. The Estate managing plan don’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 aimed 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 the fenced capture sites, to repair fences for agriculture crops, costs regarding assurance after deer-vehicle collisions.

The main aim of the Park deer management plan is to reduce the population density at a level of 12 deers / 100 ha ± 2.5 (which means a population of about 280 / 580 deers in Spring time).

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 respecting two-year olds for females, and 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.
Considering that during the spring census the 75% of females are pregnant, before the harvesting season we have an underestimation of the population of such a way (assuming a sex-ration for fawns of 0.5):

<table>
<thead>
<tr>
<th></th>
<th>2001</th>
<th>2002</th>
<th>2003</th>
<th>2004</th>
<th>2005</th>
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<tbody>
<tr>
<td>males 1-2 year old</td>
<td>42</td>
<td>245</td>
<td>162</td>
<td>102</td>
<td>105</td>
</tr>
<tr>
<td>males 2-4 year old</td>
<td>68</td>
<td>89</td>
<td>81</td>
<td>102</td>
<td>69</td>
</tr>
<tr>
<td>males over 4 year old</td>
<td>57</td>
<td>75</td>
<td>36</td>
<td>80</td>
<td>54</td>
</tr>
<tr>
<td>femal. over 2 year old</td>
<td>559</td>
<td>720</td>
<td>938</td>
<td>723</td>
<td>587</td>
</tr>
<tr>
<td>femal. 0-1 year old</td>
<td>444</td>
<td>408</td>
<td>635</td>
<td>403</td>
<td>352</td>
</tr>
<tr>
<td>males 0-1 year old</td>
<td>444</td>
<td>408</td>
<td>635</td>
<td>403</td>
<td>352</td>
</tr>
</tbody>
</table>

(Tab. 2)

In Tab. 2 the data regarding the year 2005 will be used as control data in comparison with the results deriving by the projection of the Leslie Matrix to predict the number of animals in the future.

The following matrices are been developed respectively for females and for males:

\[
\begin{pmatrix}
0 & 0.75 \\
0.5 & 0.5 \\
\end{pmatrix} \quad \text{for females} \quad \begin{pmatrix}
0 & 0 & 0 & 0 \\
0.5 & 0 & 0 & 0 \\
0 & 0.5 & 0 & 0 \\
0 & 0 & 0.6 & 0.5 \\
\end{pmatrix} \quad \text{for males}
\]

To have a control check of our Leslie Matrix we can use the number of animals estimated in 2004 as a unitary vector that will time the corresponding matrix for females and for males; in this way we obtain other two unitary vectors representing the possible number of animals that we should count in 2005. As we have this information by field data. Comparing these results, we observed that there is a difference of about 50 animals, so we can accept the parameters used in the developed of our matrices.

Additional topics concerning the biological analysis and the corresponding bio-economic considerations will be widely discussed in the next paper.
Conclusions

Bioeconomic analysis provides a powerful framework for policy development in invasive pest management. The approach explicitly links the biological components of a pest-resource system, with the economic imperatives that drive its management (Stocker, 1983). Bioeconomic analysis can be applied to the management of commercially important invasive species, as well as environment and threats. In the former instance, the monetary benefits and costs associated with pest control can be contrasted directly. In the latter instance, analysis focused on identifying pest management strategies that maximize environmental or biodiversity benefits, or achieves some specified level of mitigation at least cost (Fig. 7).

(Fig. 7 Generic decision chart indicating which type of benefit/cost analysis should be applied to particular invasive species problems)
Although the capacity to formally analyze management options for invasive species is clearly of benefit to a range of policy makers, the emphasis that bioeconomic analysis places on the development of conceptual, analytical, and/or simulation models, produces a range of collateral benefits. These includes:

* a structured analysis of the problem – model development requires a clear articulation of the impacts a pest species is thought to have, who the beneficiaries of the control are, what the consequences of not controlling the pest will be;

* a review of existing data and information – model development involves a formal analysis of critical information gaps exist concerning the pest, its control, and its impacts. As such, bioeconomic can be used to prioritise research questions and identify critical monitoring points in the management systems;

* a tool for integrating new information and data as they come to hand – the development of bioeconomic models provides a framework for integrating new information and data as it comes to hand. By ensuring that the best available information is always available to managers and policy makers, these models become the primary mechanism for ensuring best practice management and decision-making. Models can also provide an “institutional memory” of why particular policy positions were adopted, or management decisions made.

The management scenarios for a fallow deer population as we show in this work could be used to implement and develop new environmental policies necessary to address the impacts of this species (and many more others) on our environment balance, and the benefits deriving by the presence of this species, considering all the variables acting (Choquenot and Hone, 2002).
References


