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**Assessing and modelling socio-ecological
interactions of sacred forests in northwestern
Greece**

Valentino Marini Govigli

Supervisor:

John M. Halley

Members:

John R. Healey

Despoina Vokou

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ΠΑΝΕΠΙΣΤΗΜΙΟ ΙΩΑΝΝΙΝΩΝ

ΣΧΟΛΗ ΕΠΙΣΤΗΜΩΝ ΥΓΕΙΑΣ

ΤΜΗΜΑ ΒΙΟΛΟΓΙΚΩΝ ΕΦΑΡΜΟΓΩΝ ΚΑΙ ΤΕΧΝΟΛΟΓΙΩΝ

ΔΙΔΑΚΤΟΡΙΚΗ ΔΙΑΤΡΙΒΗ

**Ανάλυση και μοντελοποίηση
κοινωνικο-οικολογικών
αλληλεπιδράσεων στα ιερά δάση της
βορειοδυτικής Ελλάδας**

Valentino Marini Govigli

Επιβλέπων:

John M. Halley

Μέλη:

John R. Healey

Δέσποινα Βώκου

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1. Επιβλέπων: John Maxwell Halley, Καθηγητής, Τμήμα Βιολογικών Εφαρμογών και Τεχνολογιών, Πανεπιστήμιο Ιωαννίνων, Ελλάδα
2. Μέλος: Jenny L. Wong, Λέκτορας, Τμήμα Περιβάλλοντος, Φυσικών Πόρων και Γεωγραφίας, Πανεπιστήμιο Bangor, Ηνωμένο Βασίλειο
3. Μέλος: John R. Healey, Καθηγητής, Τμήμα Περιβάλλοντος, Φυσικών Πόρων και Γεωγραφίας, Πανεπιστήμιο Bangor, Ηνωμένο Βασίλειο

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1. Επιβλέπων: John Maxwell Halley, Καθηγητής, Τμήμα Βιολογικών Εφαρμογών και Τεχνολογιών, Πανεπιστήμιο Ιωαννίνων, Ελλάδα
2. Μέλος: John R. Healey, Καθηγητής, Τμήμα Περιβάλλοντος, Φυσικών Πόρων και Γεωγραφίας, Πανεπιστήμιο Bangor, Ηνωμένο Βασίλειο
3. Μέλος: Δέσποινα Βώκου, Καθηγήτρια, Τμήμα Βιολογίας, Αριστοτέλειο Πανεπιστήμιο Θεσσαλονίκης, Ελλάδα

Μέλη επταμελούς εξεταστικής επιτροπής από τη Γ.Σ.Ε.Σ.: 280/17/7/2020:

1. John Maxwell Halley, Καθηγητής, Τμήμα Βιολογικών Εφαρμογών και Τεχνολογιών, Πανεπιστήμιο Ιωαννίνων, Ελλάδα
2. John R. Healey, Καθηγητής, Τμήμα Περιβάλλοντος, Φυσικών Πόρων και Γεωγραφίας, Πανεπιστήμιο Bangor, Ηνωμένο Βασίλειο
3. Δέσποινα Βώκου, Καθηγήτρια, Τμήμα Βιολογίας, Αριστοτέλειο Πανεπιστήμιο Θεσσαλονίκης, Ελλάδα
4. Αναστάσιος Τρογκάνης, Καθηγητής, Τμήμα Βιολογικών Εφαρμογών και Τεχνολογιών, Πανεπιστήμιο Ιωαννίνων, Ελλάδα
5. Bart Muys, Καθηγητής, Τμήμα Δασών, Φυσικού Περιβάλλοντος και Τοπίου, Πανεπιστήμιο KU Leuven, Βέλγιο
6. Άρης Κυπαρίσσης, Αναπληρωτής Καθηγητής, Τμήμα Γεωπονίας, Φυτικής Παραγωγής και Αγροτικού Περιβάλλοντος, Πανεπιστήμιο Θεσσαλίας, Ελλάδα
7. Norman Dandy, Senior Research Fellow και Διευθυντής του Sir William Roberts Centre for Sustainable Land Use, Πανεπιστήμιο Bangor, Ηνωμένο Βασίλειο

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Mr. Valentino Marini Govigli, date of PhD application: 17-01-2014

Members of the Supervising Committee (16-01-2015):

1. Supervisor: John Maxwell Halley, Professor, Department of Biological Applications and Technology, University of Ioannina, Greece
2. Member: Jenny L. Wong, Honorary Lecturer, School of Environment, Natural Resources and Geography, Bangor University, UK
3. Member: John R. Healey, Professor, School of Environment, Natural Resources and Geography, Bangor University, UK

Subject of the thesis: Assessing and modelling socio-ecological interactions of sacred forests in northwestern Greece; defined on 16-01-2015

Members of the Supervising Committee (following reorganisation with date 15-06-2020):

1. Supervisor: John Maxwell Halley, Professor, Department of Biological Applications and Technology, University of Ioannina, Greece
2. Member: John R. Healey, Professor, School of Environment, Natural Resources and Geography, Bangor University, UK
3. Member: Despoina Vokou. Professor, Department of Biology, Aristotle University of Thessaloniki, Greece

Members of the examination board: (17-07-2020)

1. John Maxwell Halley, Professor, Department of Biological Applications and Technology, University of Ioannina, Greece
2. John R. Healey, Professor, School of Environment, Natural Resources and Geography, Bangor University, UK
3. Despoina Vokou, Professor, Department of Biology, Aristotle University of Thessaloniki, Greece
4. Anastassios Troganis, Professor, Department of Biological Applications and Technology, University of Ioannina, Greece
5. Bart Muys, Professor, Division of Forest, Nature and Landscape, KU Leuven University, Belgium
6. Aris Kyparissis, Associate Professor, Department of Agriculture Crop Production and Rural Environment, University of Thessaly, Greece
7. Norman Dandy, Senior Research Fellow, Centre for Sustainable Land Use, University of Bangor, UK

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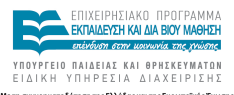
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Abstract

The purpose of this thesis is to disentangle the relative influence of ecological and social processes in the composition, spatial patterning and temporal extent of Sacred Natural Sites (SNS). SNSs are natural places with spiritual and religious significance for local communities. They are peculiar coupled socio-ecological systems arising from long-lasting interactions between ecological processes and anthropogenic influences. SNSs have been recently recognized as providers of crucial nonmaterial benefits related to human-nature relationships. However, how different ecological and social processes interact in time and space influencing their formation remains a largely unexplored topic. This is especially relevant, as SNSs face increasing risks of degradation due the cultural and demographic changes occurred in the past decades (e.g. depopulation of rural areas, knowledge loss of traditional management practices).

We use ecological and social data, collected on a sample of sacred forests in Northwestern Greece, to: (i) understand the main socio-ecological processes that led to the sacred forests establishment, (ii) identify spatial (i.e. forests extent and boundaries) and temporal (i.e. forests' establishments) patterns across sacred forests; (iii) model the spatial extent of sacred forests over time to quantify how social processes impact the forest expansion. We do this through an interdisciplinary methodology, which combines diverse set of data stemming from different sources (data on forest structure and vegetation, tree rings data, ethnographic data) integrated and analysed in the thesis using both quantitative and qualitative techniques (statistical analysis, spatial analysis, environmental modelling).

According to the findings of this thesis, we conclude that it is possible to disentangle the relative influence of ecological and social processes in the composition, spatial patterning and temporal extent of SNS. In the context of sacred forests in northwestern Greece, results suggest that traditional management practices and social beliefs are main

drivers in defining the spatial structure of sacred forests, and in preserving them as time passes. Unfolding the role of these processes is fundamental for understanding how the historical change in cultural practices related to management of sacred sites has determined the trajectory of their vegetation composition and function. This analysis provides the first multidisciplinary assessment of SNSs role in a changing society, giving clearer perspectives on how to design conservation practices tailored to SNSs facing cultural abandonment.

Περίληψη

Σκοπός αυτής της διατριβής είναι να διερευνήσει τη σχετική επίδραση των οικολογικών και κοινωνικών διαδικασιών στη σύνθεση, τη χωρική διαμόρφωση και το μέγεθος διαχρονικά των Ιερών Φυσικών Τόπων (ΙΦΤ). Ιεροί Φυσικοί Τόποι υπάρχουν σε πολλές περιοχές του κόσμου και έχουν πνευματική και θρησκευτική σημασία για τις τοπικές κοινότητες. Πρόκειται για ιδιόμορφα, συζευγμένα κοινωνικο-οικολογικά συστήματα που διαμορφώνονται από μακροχρόνιες αλληλεπιδράσεις μεταξύ οικολογικών διαδικασιών και ανθρωπογενών επιδράσεων. Οι ΙΦΤ αναγνωρίστηκαν πρόσφατα ως πάροχοι ζωτικών μη-υλικών ωφελειών που συνδέονται με τη σχέση ανθρώπου-φύσης. Ωστόσο, το πώς οι διάφορες οικολογικές και κοινωνικές διαδικασίες αλληλεπιδρούν στο χρόνο και στο χώρο επηρεάζοντας τη διαμόρφωσή τους παραμένει σε μεγάλο βαθμό ανεξερεύνητο θέμα. Αυτό είναι ιδιαίτερα σημαντικό, καθώς πολλοί ΙΦΤ αντιμετωπίζουν υψηλούς κινδύνους υποβάθμισης λόγω των πολιτισμικών και δημογραφικών αλλαγών που σημειώθηκαν τις τελευταίες δεκαετίες (π.χ. ερήμωση αγροτικών περιοχών, απώλεια γνώσεων παραδοσιακών πρακτικών διαχείρισης, κ.ά). Στην προσπάθεια να συμβάλουμε στη διερεύνηση αυτού του θέματος, επιλέξαμε προς μελέτη έναν αριθμό ιερών δασών στη βορειοδυτική Ελλάδα και συλλέξαμε και αναλύσαμε δεδομένα τόσο οικολογικού όσο και κοινωνικού χαρακτήρα.

Στο πρώτο Κεφάλαιο, εκτιμάμε και αξιολογούμε τις σχετικές επιπτώσεις των φυσικών περιβαλλοντικών παραγόντων και των καθεστώτων διαχείρισης στη δομή και τη σύνθεση ομάδων ξυλωδών φυτών σε έξι ιερά δάση στην περιοχή της Ηπείρου. Προς τούτο κάναμε στο πεδίο δασική απογραφή και προχωρήσαμε σε προσδιορισμό της αφθονίας των ειδών που συμμετέχουν και της εξ αυτής κατάταξής τους, ανάλυση συστάδων και υπολογισμό δεικτών ανομοιότητας. Τα αποτελέσματά μας δείχνουν ότι η αφθονία των ειδών και η δομή των δασών ποικίλλουν σε μεγάλο βαθμό μεταξύ περιοχών, με αξιοσημείωτες διαφορές μεταξύ του ορόφου της κομοστέγης και του υπο-ορόφου σε ορισμένα ιερά δάση, που μπορεί να συνδέονται με ανθρωπογενείς

διαταραχές ή επέκτασή τους σε εγκαταλειμμένα χωράφια κατά το παρελθόν. Αναγνωρίσαμε τέσσερις κύριες ομάδες ξυλωδών φυτών που σχετίζονται ποικιλοτρόπως με περιβαλλοντικές μεταβλητές και μεταβλητές διαχειρίσιμες δασών. Το βασικό μας εύρημα είναι ότι υπάρχει διαφοροποίηση στη σύνθεση των ειδών στα ιερά δάση που συνδέεται, εκτός από τις βασικές περιβαλλοντικές μεταβλητές (όπως η έκθεση της θέσης όπου εμφανίζονται), και με την ένταση και τον τύπο των καθεστώτων απαγόρευσης των δασών (από περιοχές με αυστηρά καθεστώτα απαγόρευσης μέχρι περιοχές όπου επιτρέπεται η βόσκηση). Συγκεκριμένα, οι ιστορικές απαγορεύσεις (κυρίως ο έλεγχος της βόσκησης) είναι ένας από τους κύριους καθοριστικούς παράγοντες της τρέχουσας δομής των ιερών δασών.

Στο δεύτερο και τρίτο κεφάλαιο, εξετάζουμε τη χωροχρονική δυναμική των κοινωνικών και οικολογικών διαδικασιών που επηρεάζουν τα ιερά δάση. Ειδικότερα, στο δεύτερο εστιάζουμε στον χρόνο. Χρησιμοποιούμε τεχνικές δενδροχρονολόγησης για να εκτιμήσουμε την ηλικία των δέντρων σε πέντε ιερά δάση και ακολούθως να συνδέσουμε διακριτά κύματα αναγέννησης της κομοστέγης των δέντρων με τοπικές κοινωνικο-οικονομικές αλλαγές. Υπολογίζουμε ότι η συστάδα δέντρων με τις μεγαλύτερες κομοστέγες από τα πέντε δάση είναι περίπου 225-250 ετών. Αυτό το εύρημα ταιριάζει με τις διαθέσιμες ιστορικές ανασυγκροτήσεις. Επιπλέον, εντοπίζουμε δύο κύριες κοινές περιόδους αναγέννησης στα πέντε μελετημένα ιερά δάση, οι οποίες αντιστοιχούν στο δεύτερο μισό του 19ου αιώνα και στο δεύτερο μισό του 20ου αιώνα, που συμπίπτουν με περιόδους πληθυσμιακής συρρίκνωσης, αναταραχών και ανασφάλειας. Αυτό το αποτέλεσμα επιβεβαιώνει την κύρια υπόθεση της διατριβής ότι άνθρωπος και οικολογία των ιερών δασών στη βορειοδυτική Ελλάδα βρίσκονται σε αλληλεπίδραση.

Στο τρίτο κεφάλαιο εστιάζουμε στο χώρο και μελετάμε την επίδραση των κοινωνικο-οικολογικών διαδικασιών στην έκταση των ιερών δασών. Εστιάζουμε την ανάλυσή μας στα εκάστοτε όρια των ιερών δασών. Στόχος μας είναι να κατανοήσουμε πώς

διαμορφώνονται τα όρια, λαμβάνοντας υπόψη το ρόλο που διαδραματίζουν οι αντιλήψεις αλλά και τα ταμπού των ανθρώπινων κοινωνιών. Διαπιστώνουμε ότι η χαρτογράφηση της ακριβούς έκτασης ενός ΙΦΤ είναι μια πολύπλοκη διαδικασία λόγω της ασαφούς και διάχυτης φύσης των ορίων τους που εξαρτώνται από πολλαπλότητα παραγόντων, οι οποίοι αλλάζουν δραματικά με την πάροδο του χρόνου. Παρ' όλα αυτά, καταφέραμε να προσδιορίσουμε ποια χαρακτηριστικά ανήκουν στον πυρήνα της ιερής περιοχής (π.χ. η ύπαρξη και θέση εκκλησίας, θέση των παλαιότερων δενδροκοινοτήτων) και ποια άλλα είναι περιφερειακά στους ιερούς τόπους.

Στο τελευταίο κεφάλαιο, αναπτύσσουμε ένα μοντέλο που να μπορεί να εκτιμήσει πώς αλλάζει η έκταση που καταλαμβάνουν τα ιερά δάση με βάση τους ποικίλους κοινωνικο-οικολογικούς παράγοντες και περιορισμούς που επιβάλλουν. Στόχος είναι να προσδιοριστεί σε ποιο βαθμό οι κοινωνικο-οικονομικές διεργασίες (δηλαδή οι δημογραφικές και οι διαχειριστικές αλλαγές) μπορούν να επηρεάσουν τη χωρική έκταση των ιερών δασών, η επέκταση των οποίων θα καθοριζόταν σε διαφορετική περίπτωση από οικολογικές διαδικασίες αποκλειστικά. Οικολογικά και κοινωνικά δεδομένα από τις προηγούμενες εφαρμογές χρησιμοποιήθηκαν για την ανάπτυξη ενός μοντέλου κυψελικού αυτόματου (Cellular Automata) που ενσωματώνει μια διαδικασία θορύβου $1 / f$ για τη μοντελοποίηση της περιβαλλοντικής μεταβλητότητας στο χώρο και το χρόνο. Αυτό το χωροχρονικό μοντέλο μας επέτρεψε να προσομοιώσουμε την επέκταση στο χρόνο ενός ιερού δάσους κάτω από διαφορετικές κοινωνικές διαδικασίες (ροές πληθυσμού, αλλαγές διαχείρισης, ένταση ισχύος ταμπού και καθεστώτων απαγόρευσης). Τα αποτελέσματα των προσομοιώσεων δείχνουν ότι δύο κύριες διαδικασίες, η μείωση του πληθυσμού και η αλλαγή των πρακτικών διαχείρισης, είναι κρίσιμοι κοινωνικοί παράγοντες, καθοριστικοί για τον τρόπο με τον οποίο το περιβάλλον αντιδρά, περιορίζοντας ή αυξάνοντας ανάλογα την επιφάνεια του δάσους. Συγκεκριμένα, η πληθυσμιακή συρρίκνωση συνέβαλε σε αύξηση μόνο 30% της επιφάνειας του ιερού δάσους στο περιβάλλον διαχειριζόμενο τοπίο του, ενώ η συνδυασμένη επίδραση της δημογραφικής κατάπτωσης και των διαχειριστικών αλλαγών

επιδεινώνει τη βλάστηση που γεμίζει τις εξωτερικές περιοχές του ιερού δάσους του Γρεβενιτίου κατά 15% επιπλέον. Το μοντέλο που αναπτύξαμε αποδείχθηκε χρήσιμο εργαλείο για την προσομοίωση της επίδρασης των μεταβαλλόμενων ιστορικών σεναρίων στην έκταση της βλάστησης του ιερού δάσους βάσει των διαθέσιμων δεδομένων. Πιστεύουμε ότι η εφαρμογή του μπορεί να επεκταθεί σε άλλα πολιτισμικά τοπία και ΙΦΤ για τη διακρίβωση/επικύρωση προφορικών αφηγήσεων και τοπικών ιστορικών δεδομένων.

Σύμφωνα με τα ευρήματα αυτής της διατριβής, καταλήγουμε στο συμπέρασμα ότι είναι δυνατόν να προσριοριστεί η σχετική επίδραση των οικολογικών και κοινωνικών διαδικασιών στη σύνθεση, τη χωρική διαμόρφωση και την έκταση διαχρονικά των ΙΦΤ (υπό την προϋπόθεση ότι είναι διαθέσιμο ή μπορεί να συλλεχθεί ένα σύνολο απαραίτητων οικολογικών και κοινωνικών δεδομένων). Χρησιμοποιώντας τα ιερά δάση στη βορειοδυτική Ελλάδα ως περιοχή μελέτης, αναγνωρίσαμε πώς αυτά εξελίσσονται δυναμικά στο χώρο ως αντίδραση σε μια σειρά κοινωνικο-οικολογικών διαδικασιών, οι οποίες επηρεάζουν και την εσωτερική σύνθεση και τη δομή τους. Παρόλο που οι οικολογικοί παράγοντες διαδραματίζουν έναν κρίσιμο ρόλο στη διαμόρφωση της δυναμικής των θέσεων που καταλαμβάνουν, διαπιστώσαμε ότι ορισμένες κοινωνικές διαδικασίες είναι επίσης ζωτικής σημασίας για την ποιότητα των ΙΦΤ ως προς τη σύνθεση και τη χωρική διαμόρφωσή τους. Ανάμεσά τους συγκαταλέγονται κανόνες απαγόρευσης και πολιτισμικές ιεροπραξίες, διαχείριση της εξωτερικής ζώνης του δάσους και τάσεις του πληθυσμού. Η αποκάλυψη του ρόλου αυτών των διαδικασιών είναι θεμελιώδης για την κατανόηση του τρόπου με τον οποίο η ιστορική αλλαγή στις πρακτικές που σχετίζονται με τη διαχείριση των ιερών τόπων έχει καθορίσει τη δυναμική τους, ως προς τη διαμόρφωση της βλάστησης και τη λειτουργίας τους. Αυτή η ανάλυση αποτελεί την πρώτη διεπιστημονική αξιολόγηση του ρόλου των ΙΦΤ σε μια μεταβαλλόμενη κοινωνία και μπορεί να συμβάλει στο σχεδιασμό πρακτικών διαχείρισης προσαρμοσμένων στη διατήρηση ΙΦΤ που αντιμετωπίζουν πολιτισμική εγκατάλειψη.

CHAPTER I

Introduction and theoretical background: setting the scene

I.1 Theoretical background: cultural landscape and sacred natural sites

Contemporary rural landscapes are the consequence of long-lasting interactions between ecological processes and anthropogenic influences. Distinct landscapes emerge from these processes, which reflect *“ideas, customs, and social behaviour of a particular people or societies”*, i.e. what we conceive as human cultures (OED, 2020) and are conceptualised as Cultural Landscapes. Some examples of widely recognized cultural landscapes include: Uluru Mountain in Australia (also called Ayers Rock), sacred to aboriginal peoples; the Alto Douro Region in Portugal, with a system of vineyards which have produced wine for more than 2000 years; and the ancient system of rice terraces present in the Philippine islands and other Asian countries (Fowler, 2002).

The notion of cultural landscape was firstly formulated in 1906 by the geographer Otto Schlüter, as the consequence of *human culture acting on the primeval landscape through the medium of time* (Rapoport, 1992). Nonetheless this concept was implicitly employed by painters from the Renaissance onwards who were increasingly interested in depicting an idyllic idealized vision of nature and their relationship with people (Fig. I.1). It was then officially formalized by Carl O. Sauer in 1925, when he defined cultural landscape as a social construct arising from the moulding of natural spaces by cultural processes: *“The cultural landscape is fashioned from a natural landscape by a culture group. Culture is the agent, the natural area is the medium and the cultural landscape is the result.”* (Sauer, 1925). With this definition it clearly appears how, one hundred years ago, the primary responsible for shaping landscapes were assumed to be people and communities who mould nature according to different cultural practices.

It is only in recent times that the cultural landscape notion has gradually shifted towards a more balanced and reciprocal people-nature relationship. The 1972 UNESCO convention on World Cultural and Natural Heritage defines cultural heritages as the *“combined works of nature and man [...] which are of outstanding universal value from the*



Figure I.1: Two examples of Landscape Painting depicting diverse idyllic nature sceneries and the way primeval societies interact with it. Left side, Claude Lorrain, Pastoral landscape with a Mill (1634); right side, Thomas Cole, The Course of Empire: The Arcadian or Pastoral State (1833-36).

historical, aesthetic, ethnological or anthropological point of view" (UNESCO, 1972). In 1992, the World Heritage Convention became the first international legal treaty with the aim of recognizing, protecting and conserving cultural landscapes across the world (UNESCO, 2020b). In 2007, the first definition of Coupled Human and Natural Systems (CHANS) was proposed to recognise the reciprocal interaction of natural and social processes on ecosystems: *"Coupled human and natural systems are integrated systems in which people interact with natural components. [...] People and nature interact reciprocally and form complex feedback loops"* (Liu et al., 2007). Cultural landscapes are thus directly and dynamically dependent both on natural and social processes. They are complex constructs in which human and natural components mutually interact with different intensity and at different spatial and temporal scales.

The value of cultural ecosystem services has been recently recognized as important nonmaterial benefits people and communities obtain from their surrounding landscape. The 2005 Millennium Ecosystem Services Assessment lists ten such values, including: cultural diversity, spiritual and religious values, knowledge systems, educational values, inspiration, aesthetic values, social relations, sense of place, cultural heritage values, and recreation and ecotourism (Overpeck et al., 2013). More recently, the formulation of Nature Contribution to People by the Intergovernmental Platform on

Biodiversity and Ecosystem Services (IPBES), provided a stronger recognition of the role of knowledge systems and cultural contexts in human-nature relationships (Pascual et al., 2017). Despite the interaction of nature and society has been deeply studied in urban contexts and rural-urban interfaces (Salvati and Serra, 2016; Schmitz et al., 2018), few studies (e.g. Arnaiz-Schmitz et al., 2018) have attempted investigating how different ecological and social processes interact influencing the formation of cultural ecosystem services and landscapes.

Spiritual and religious practices indicate the central role played by terrestrial and aquatic landscapes in the majority of major religions and cultures (FAO, 2020). This finds particular expression in the identification of specific places, often associated with distinct natural features which are deemed sacred. These *“areas of land or water having special spiritual significance to peoples and communities”* are generally indicated as Sacred Natural Sites (SNS) (Verschuuren, 2010; Wild and McLeod, 2008; Oviedo et al., 2005). SNS are not constrained to any particular region nor culture. Bhagwat and Rutte (2006) in their extensive review, reported 33 examples of SNS occurring in ten different biomes on all continents.

SNSs are dramatically site-specific and differ in their geomorphology, dimension, and spiritual status (Mallarach, 2012). SNSs include mountains, rivers and lakes, forests, caves, and islands (Schaaf and Lee, 2006), and they range from being single trees, ponds, or rocks (i.e. the Bodhi Tree in India, worshipped as ancient fig tree under which the Buddha reached enlightenment; Nugteren, 1995), to entire mountain ranges (i.e. the Grand Canyon National Park in Arizona, sacred ground for several Native American tribes; Ross-Bryant, 2005; Fig. 1.2). Unique is also the reason behind their intrinsic spirituality. SNSs can be homes of gods and the ancestors' spirits (i.e. several sacred mountains of Japan believed to be personifications of Shinto spirits or gods; Byrne, 2010). In other cultures, SNSs are important spaces used for ceremonies, prayers and meditation (e.g. numerous pilgrimage sites in Europe such as Mount Athos in Greece;

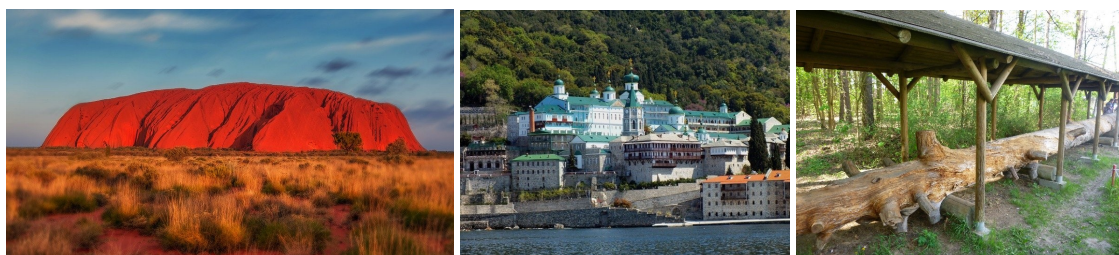


Figure I.2: Examples of Sacred Natural Sites. From the left: Uluru Mountain in Australia, sacred to the Pitjantjatjara Anangu, Aboriginal people of the area; Mount Athos an important centre of Eastern Orthodox monasticism in Greece; The sacred pine of Suprasl, Poland, which is venerated, stocked, and safeguarded even after it naturally died. © Pixabay and Marini Govigli V.

Papayannis, 2012). Sometimes they are interlinked with physical cultural artefacts, as churches, sanctuaries, temples or burial grounds, and they might be inherited from previous civilizations (i.e. the Majella National Park in Central Italy were sacred sites were persistently preserved and inherited by subsequent civilizations over more than 800,000 years; de Waal, 2012).

Laws, rules and regulations for ethical and moral behaviour often regulate access to SNSs (Oviedo et al., 2005). These formal rules or informal norms aim to preserve the site integrity and respect for its inherent spirituality. In certain cases all access to site is restricted (e.g. the ban implemented in October 2019 prohibiting access to Uluru mountain; NYT, 2019. Also, Mt Kailas, sacred to Hindus, Buddhists, Jains and Bon, which is closed to mountaineers; Jornet, 2016), while in others access is limited to specific groups such as devotees or pilgrims. In others access is permitted but certain management practices are forbidden or recommended (i.e. in Madagascar where the survival of many communities depends almost exclusively on sacred forests during periods of drought; Oviedo et al., 2005). These rules and regulations are in many cases unknown or kept secret to outsiders of the community.

Strong adherence to the SNSs internal governance rules have allowed many sites to remain largely untouched and have alleviated human pressure over long periods of time. This resulted in many SNSs being sanctuaries of flora and fauna biodiversity and

remnants of rare vegetation types otherwise degraded by stronger human pressure (Dudley et al., 2009; Wild and McLeod, 2008). This indicates how SNSs have increased the survival chances of many rare natural habitats across the world. Additionally SNSs are also important sources of economic subsistence for many communities, providing important tangible resources (e.g. fodder, medicinal plants, game meat, etc.). Another prominent characteristic of SNSs, is the role played by local communities in the management, protection and conservation of such places. Local guardians can be formed by representatives of religious institutions, local community leaders, or more dispersed groups (Wild and McLeod, 2008). As Dudley et al. (2005) report, SNS are probably the oldest protection system for habitats on the planet.

Because SNSs have multiple cultural, economic, and biodiversity values, many initiatives have thus started to better recognize their potential (e.g. the Sacred Natural Site initiative; www.sacrednaturalsites.org). However, as the world demographic and cultural contexts change and especially with urbanization processes which depopulate rural areas and dissipate traditional management practices (Lieskovský et al., 2015), SNSs face increasing risks of degradation (Overpeck et al., 2013). Meanwhile, stresses arising from rural overpopulation (as in many developing countries) might promotes overexploitation of certain SNSs, due to excessive use and human pressure (Wild and McLeod, 2008). As a result, an assessment of their role in a changing society becomes urgent for scientists and practitioners.

Sacred forests and groves are a sub-set of SNSs, being sacred sites made up of trees, from small groves to larger forested areas. As, SNSs, they can be found in many different cultures around the world. The literature mentions sacred forests in countries such as: India, Tibet, China, Japan, Indonesia, Ethiopia, Morocco, Tanzania, Australia, Estonia, United Kingdom, Italy, and Greece (Xu et al., 2006; Daye and Healey, 2015; Verschuuren, 2010; Frosch et al., 2017; Zhuang and Corlett, 1997; Castro and Aldunate, 2003; Rose et al., 2003; Chandran and Hughes, 2000; Ylhäisi, 2006; Frascaroli et al., 2019;

Mallarach et al., 2012). Sacred groves vary markedly in size and management regime, depending on whether they serve practical (secular) needs or are sites for worship and rites (Dudley et al., 2005).

In sacred forests it is often the local community that acts as the custodian of the site, managing and conserving its cultural, spiritual and environmental value (Wild and McLeod, 2008). In many sacred forests, nature conservation and productive management practices are often combined. The custodial groups usually together with local religious authorities define specific management rules, including duties and prohibitions on forest use. Such rules can vary from site to site to reflect local needs (Frosch et al., 2017; Stara et al., 2016). Governing rules, together with socio-economic settings and environmental and climatic patterns, contribute to sacred forests inter- and intra-site diversity. Nevertheless, despite this diversity, these forests often share common features. Bhagwat and Rutte (2006) showed that such sites are often situated in agricultural landscapes and in close proximity to settlements. Jäckle et al. (2013) found that sacred forests in Northern Morocco are mainly located on elevated and prominent areas. As expected, sacred sites are observed to have higher biodiversity levels relative to neighbouring habitats (Wild and McLeod, 2008; Dudley et al., 2009; Gao et al., 2013; Avtzis et al., 2018). However, different conclusions might be drawn when the sacred forest is small in size (Garcia et al., 2006), when the condition of the surrounding landscape matrix becomes crucial for the biodiversity of the groves (Daye and Healey, 2015).

I.2 Problem statement

The purpose of this thesis is to disentangle the relative influence of ecological and social processes in the composition, spatial patterning and temporal extent of SNSs. While SNSs have been recognized as providers of crucial nonmaterial benefits related to human-nature relationships (Wild and McLeod, 2008), how different ecological and social processes interact in time and space influencing their establishment remains a

largely unexplored topic. This is especially relevant, as SNSs face increasing risks of degradation due the cultural and demographic changes occurred in the past decades (e.g. depopulation of rural areas, abandonment of traditional management practices). In particular, sacred forests in Europe and in the Mediterranean are threatened by negligence and knowledge loss, as large numbers of people in the 20th century have moved away from rural areas abandoning traditional dwellings and lifestyles. Such abandonment has an impact on the whole landscape with fields and pastureland left derelict allowing widespread forest encroachment (Blondel et al., 2010).

While the roles of SNS and sacred forests in preserving traditional knowledge and management practices (cultural values) and environmental resources (natural values) have been both widely explored by authors in the literature (e.g. Oviedo et al., 2005; Dudley et al., 2012), certain research gaps are yet to be addressed. These include a more rigorous assessment of the magnitude and directionality of interactions between ecological and cultural processes, in order to explore which of them prevails in time and space.

De-coupling socio-ecological drivers behind SNSs formation is crucial as it allows understanding how they will evolve in the future. This is especially relevant because humane-nature relationships are drastically changing (Hung, 2018). If intangible cultural values fade, then the socio-cultural construct behind the formation and preservation of a given cultural landscape is also changing. This process can thus undermine the effectiveness of landscape management practices that do not consider the evolution of spiritual and religious values through time. This analysis would then be useful for national policy makers and conservation managers of SNSs as it will allow identifying the extent to which faiths and traditional management practices can support natural conservation approaches (Dudley et al., 2009), and properly manage scenarios in which religious practices fall into disuse (Anyinam and Kalipeni, 1999).

I.3 Research questions and objectives

Table 1 provides an overview of the research questions guiding this thesis. The background proposition of this research is that existing and former management practices and prohibition bans of sacred forests affect their ecological structure and dynamics both in space and time. In particular, this proposition can be sub-articulated into the following **research questions**:

- Can we identify the extent to which variation in species composition in sacred forests can be linked to former management practices and prohibition regimes?
- Can we link waves of tree recruitment to common regional socio-economic changes in time?
- Can we map spatially the forests extent and identify how different natural, cultural, perceptive, and geophysical factors affect the composition of their boundaries?
- Can we model quantitatively how ecological processes of forest expansion and socio-economic processes (i.e. demographic and management changes) determined across time the spatial extent of sacred forests?

The general objective of this research is to demonstrate that ecological and social processes interact reciprocally in forging SNS and sacred forests taking a selection of sacred forests in northwestern Greece as case studies. This thesis is a contribution to existing knowledge on coupled socio-ecological systems, with the aim of improving understanding of how social and ecological processes coexist and together create anthropogenic landscapes. We finally aim at assessing whether, conceptualizing SNSs using the Socio-Ecological Systems (SES) perspective, bring us innovative viewpoints on how these sites work, with relevant policy implications. Only few studies (e.g. Colding et al., 2003) have so far attempted to conceptualize sacred forests through SES (Box 1).

Box 1 - Socio-Ecological Systems

Berkes and Folke (1998) define SESs as *linked systems of people and nature* and emphasise the complex interplay of ecological, social, economic and institutional realms, both in time and space, when dealing with ecosystems (Folke et al., 2005). The concept of SESs has been extensively used in literature as a methodological tool to study interlinked environment and human systems, which rarely can be understood in isolation due to their multiple and reciprocal feedback loops (Cote and Nightingale, 2012; Anderies et al., 2004). Understanding these systems requires an integrated holistic view of their component and their interrelationships (Carson and Peterson, 2016). Thus multi and inter disciplinarily tools are advocated to facilitate the consolidation of ecological, economic and social science techniques and theories into unique frameworks of study (Berkes and Folke, 1998). Additionally, the SES approach is particularly suitable when studying territories with long-lasting human and natural relationships, such as the ecosystems of Europe and the Mediterranean basin (FAO and Plan Bleu, 2018). Figure 3 illustrates the main dynamics around SES (Source: Carson and Peterson, 2016):

- Inter-systems interactions and feedback loops linking natural components to human well-being, and human activities to ecosystem functioning (arrows around the periphery);
- Intra-systems interactions connecting primarily biophysical, ecological, and social processes (i.e. the physiological response of forests to water, or on the other hand, institutional societal arrangements such as monetary policies) (arrows within the figure).

This figure highlights the intrinsic complexity of SESs, which depends on both biophysical and social sub-systems dynamics.

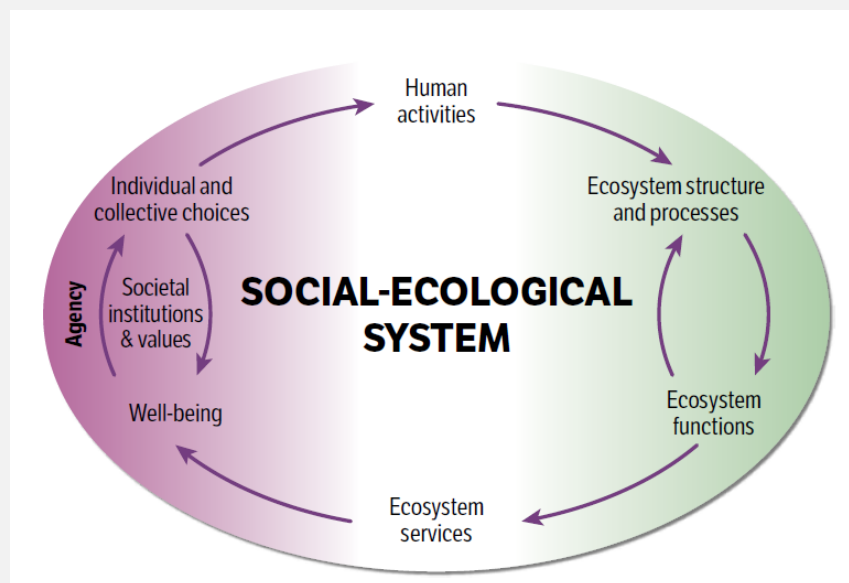


Figure I.3: A schematization of social-ecological systems and intrinsic interactions (Source: Carson and Peterson, 2016).

Table I.1: The research matrix of this thesis, identifying research questions and related methodological approaches and analyses.

Question	Hypothesis	Sub-hypothesis		Data (units) [social]	Data (units) [ecological]	Methods	Analysis	PhD Chapter
it possible to disentangle the relative influence of ecological and social processes in the composition, spatial patterning and temporal extent of SNS?	[Using six sacred forests as an example of long-lived social rules] Existing and former management practices and prohibition bans of sacred forests control their ecological structure and dynamics both in space and time	Variation in species composition across sacred forests can be attributed to former management practices and prohibition regimes	Sacred forests differ in their species composition both at site level, and between understorey and canopy vegetation. This difference can be traced back to different management regimes	Plots in six sacred forests. For each sample locations: Forest management indicators (presence of cut or pruned trees, management practices etc.)	Plots in six sacred forests. For each sample locations: Tree stand tables (species occurrence and size) for canopy and understorey Environmental variables (slope, ground cover etc.)	Forest inventory of six sacred forests	Comparison of species rank abundance distributions for both and understorey levels Bray-Curtis measure of dissimilarity to assess tree composition difference amongst plots across sites Cluster analysis based on closeness within the relative distance matrix; regression models to explore significant relationships between tree composition and environmental and management variables.	3
			Distinct waves of recruitment of canopy trees can be linked to common regional socio-economic changes	Historical village population data	Plots in five sacred forests. For each sample locations: Tree stand tables (species occurrence and size) for canopy and understorey Estimation of tree age using wood corers	Forest inventory of six sacred forests Desk and literature research	Tree ring dating of trees. Quantification of ages of three tree dominant taxa through linear regressions Qualitative assessment of correlation of canopy tree density over time derived from age/size regressions with historical village population data	4
		Former management practices together with cultural perceptions and ecological drivers all contribute in defining the spatial extent of sacred forests (i.e. their boundaries)	Boundaries of sacred forests are determined by the interplay of natural, cultural, perceptive, and geophysical factors	Interviews & archive data	Plots in three sacred forests. For each sample locations: Tree stand tables (species occurrence and size) for canopy and understorey Geographical variables (slope, altitude, aspect etc.) Estimation of tree age using wood corers	Forest inventory of three sacred forests Adaptive Cluster sampling for canopy species in three forests (census) GPS mapping of three sacred forests	Classification of boundary elements Spatial analysis to overlay ecological, geophysical, and social boundaries elements	5
			The spatial extent of sacred forests is determined by the balance of ecological processes of forest expansion and socio-economic processes (i.e. demographic and management changes)	Interviews & archive data Historical village population data	Plots in one sacred forests. For each sample locations: Tree stand tables (species occurrence and size) for canopy and understorey Literature search of ecological data (fecundity parameters, seeds production, etc)	Forest inventory of one sacred forest Simulated model of population dynamics	Cellular automata model built using available social and ecological data to simulate the ecological responses to changing socio-ecological constraints (population flows, management changes and strength of probationary rules)	6

I.4 Thesis structure

The thesis is articulated in seven chapters. Chapter I sets the scene introducing the theoretical background of the thesis and key concepts related to cultural landscape, sacred natural sites and sacred groves and presents the justification of this study, the research questions and objectives, and the thesis contents and structure. Chapter II introduces the study areas —six sacred forests in North-western Greece— and the data collection process. Chapters III, IV, V, and VI are the core of the thesis presenting four sets of analyses performed.

Chapter III: When nature meets the divine: effect of prohibition regimes on the structure and tree-species composition of sacred forests in Northern Greece

This first chapter presents the result of a systematized forest inventory conducted in order to explore what kind of woody communities dominate a sample of six different sacred forests in Northern Greece, and to identify the relative effects of physical site environment and management regimes on the structure and composition of woody plant groups in sacred mountain forests. The main goal of this study is to determine whether in sacred mountain forests there are dissimilar types of woody plant communities that reflect different prohibition regimes established in the forests.

Chapter IV: Combining tree rings and stand dynamics to identify common historical patterns of sacred forests in Northern Greece

This chapter makes use of dendrochronological techniques to date tree establishment events in five sacred forests, to validate available oral and historical narratives and identify regional patterns as an indication of the general functional role of sacred forests. The research question behind this study is verifying whether distinct waves of recruitment of canopy trees can be linked to common regional socio-economic changes (e.g. human population decline).

Chapter V: Where to draw the line: the role of cultural perceptions in boundary formation of Epirus sacred forests

The chapter presents the results of a multidisciplinary and novel mapping approach of the cultural boundaries of three sacred forests, aiming at better understanding how boundaries in cultural landscape are formed, taking into consideration also the role of peoples perception and taboos. In particular, in this study we want to investigate whether the spatial extent of sacred forests depends upon a series of multiple ecological, cultural, perceptive, and geophysical factors.

Chapter VI: Modelling the socio-ecological development of sacred forests in Epirus using a Cellular Automaton model

The focus of this chapter is to build a realistic and rigorous model framework to improve understanding of the dynamics that exist in sacred forests. A spatially explicit Cellular Automaton model was developed to test if the spatial extent of sacred forests through time is determined by the balance of ecological processes of forest expansion and socio-economic processes (i.e. demographic and management changes). In particular, we aim at estimating quantitatively the joint impacts that different social constraints (population flows, management changes and strength of taboos and probationary rules) had over the extent of one sacred forest.

A concluding chapter (Chapter VII) presents the overall conclusions of this research. Finally, a set of annexes integrated the research by providing a set of supplementary materials related to the four main chapters of the thesis. The reference list is reported in full after the concluding chapter.

CHAPTER II

Study areas and methodological approach

II.1 Sacred forest in Epirus, Northern Greece: background knowledge

The study areas of this thesis are located in the Northern part of the Epirus Region, Greece, and specifically within the mountainous communities of the local administrative units of Zagori and Konitsa municipalities. In this territory, former studies have recently unveiled and researched in detail a large network of sacred forests and groves (Avtzis et al., 2018; Stara et al., 2016). These SNSs were established through different ritual praxes, spanning from Saints dedication to community agreements, where diverse prohibition regimes coexist (e.g. controlled management, and strict prohibition of trespassing). While examples of SNSs have been identified throughout the Mediterranean basin, they are often associated to monastic settlements (in the Northern part of the Mediterranean; Mallarach et al., 2012) and Muslim burial grounds (in the Southern Mediterranean basin; Jäckle et al., 2013). No Mediterranean areas insofar have shown a variety of ritual praxes behind the SNS establishment as the ones identified in Northern Greece. This makes the network of sacred forests in Northern Greece a perfect case study for disentangling ecological and social processes behind SNSs formation, as it allows identifying recurring patterns across a variety of sites.

The Epirus Region is one of the most sparsely populated areas of Greece. Large historical migrations have deeply modified the demographic structure of the region, which shifted from a population of around 648,000 people in the mid-20th century, to 336,856 inhabitants registered in 2011 (City Population, 2020); losing approximately half of the historical population in only 60 years (Erickson, 2003).¹ In the municipalities of Zagori and Konitsa, subsequent depopulation waves due to wars and successive economic instability have impacted the socio-economic structure of mountainous rural

¹ The official population data from the census should be treated carefully as often official statistics do not match the real population living in villages, especially in rural areas. Official statistics are thus inflated by non-permanent residents there registered but living elsewhere in Greece. Retrieving the real population of rural villages in Greece requires complex ethnographic historical reconstructions (Green, 2016).

areas, where many scattered small villages were left abandoned, or with few remaining permanent residents. The effects of these processes can be immediately observed in the regional landscape rich of former agricultural areas, now left derelict allowing widespread forest expansion (Tsiakiris et al., 2009; Papanastasis, 2007).

The Northern part of the Epirus Region, host a rich flora and fauna including several endemic and endangered species (Amanatidou, 2005). Most of the area is now included in the NATURA 2000 network of protected areas, while part of it is protected as National Forest (EEA, 2019) and was designed as a National Park in 2005 (Northern Pindos National Park, 2017). The area is characterized by a Mediterranean climate of hot summers dominated by dry continental air masses and wet mild winters (Northern Pindos National Park, 2017). Rainfall in the area is intense, especially in the autumn, with about 1195 mm of rainfall per year (Papigo Weather Station, 2020). There are two to three dry summer months each year (usually July and August). Temperatures in the area corresponds to continental zones, with a high amplitude between the mean maximum (about 30.8 °C, in August) and the mean minimum (about 3.6 °C in January; Papigo Weather Station, 2020). Most of the study area is included within the Vikos - Aaos Unesco Global Geopark, a large complex of sedimentary rocks, shaped in an elevated karstic field, intensely eroded and thus characterized by deep gorges and limestone cliffs (UNESCO, 2020a).

Previous research indicates that sacred forests have existed in the area since at least the Ottoman period (1479-1912), forming small forest patches embedded in active agricultural and pastoral landscapes. These sacred forests provided several regulatory ecosystem services to their neighbouring communities, such as protective wooded buffers above villages and reserves of essential resources in times of dire need. Moreover, this variety of services is reflected in the different ritual praxes for sacred forest establishment carried out by the local communities (e.g. dedication to Saint or via excommunication threats), which in turn lead to a variety of management regimes,

ranging from strict protection to controlled management (Stara et al., 2016).

The vegetation types present in the network of sacred forests in Zagori and Konitsa mirror most of forest communities of the Mediterranean vegetation zone, including deciduous and evergreen oak forests, mixed broadleaved forests, and black pine forests (Korakis et al., 2014). However, sacred forests may differ from neighbouring forests of the same type in the structure (Korakis et al., 2014). Research on the area indicates that the sacred forests of Epirus have a conservation advantage of certain taxa when compared to matched control sites (Avtzis et al., 2018). Mature trees and deadwood debris are often more abundant in these forests, due to prohibitions on access and removal of wood. Other studies showed that the presence of senescent and decaying trees increases the site biodiversity, especially favouring abundances of species like lichens and mosses, bird species, insects and fungi (Muggia et al., 2018; Avtzis et al., 2018, 2016).

As for many sacred forests around Europe, Epirus sacred forests are now facing cultural abandonment, as people moved out from rural areas favouring larger cities throughout the region (Nitsiakos, 2016). With people moving away from the rural environment, forest encroachment has also increased on abandoned fields and pastureland in the forests proximity (Tsiakiris et al., 2019, 2009). Nonetheless, in many villages of the region sacred forests still play an important role in local folklore and the cultural identity of rural communities (Stara et al., 2016). Nowadays, thanks to a long research effort, the network of sacred forests in Northern Epirus has been included in the Greek UNESCO's Intangible Cultural Heritage Directory

(<http://ayla.culture.gr/en/iera-dasi-twn-xwriwn-tou-zagoriou-kai-tis-konitsas/>).

Epirus Sacred forests are also part of the global Sacred Natural Sites network (<https://sacrednaturalsites.org/items/the-sacred-groves-of-zagori-epirus-greece/>).

II.2 Study areas

The analyses presented in this thesis were undertaken using data collected from six sacred forests, which have been chosen out of a list of 16 sites, studied in detail in the framework of the project THALIS-SAGE (*Conservation through religion. The sacred groves of Epirus*; Box 2 and Fig. II.1). These six sites were chosen in order to cover a representative sample of the local site diversities (Table II.1). In particular, we choose an ensemble of sites which could cover: (i) different vegetation covers, (ii) elevation levels, and (iii) ritual praxes (i.e. dedication and excommunication forests) The sites are: (i) *Agios Nikolaos* in Livadakia, belonging to the village of Vitsa (village coordinates: 39°52'25.4" N, 20°44'58.0" E); (ii) *Kouri*, village of Mazi (40°02'22.3" N, 20°40'02.4" E); (iii) *Mereao*, village of Palioseli (40°02'31.1" N, 20°53'12.6" E); (iv) *Agia Paraskevi*, village of Vovoussa (39°56'14.4" N, 21°02'55.2" E); (v) *Toufa*, village of Greveniti (39°48'21.6" N, 21°00'13.6" E); and (vi) *Graditsa*, village of Kapesovo (39°53'17.2" N, 20°46'48.8" E). Table I.1 in Chapter I exemplifies how the different study areas have been integrated in the overall methodology and the thesis main research questions. Additional ecological and social data from the larger pool of the 16 Thalys-SAGE sites were also used to interpret and generalize the findings. In the following paragraphs, we present a short description of the studied forest sites².

²In the thesis we refer to the studied sacred forests using the associated village name.



Figure II.1: Location of 16 sacred forests in the Epirus region studied in the project THALIS-SAGE. The six forests in red boxes are the study areas of this thesis. ©Adapted from Stara et al., 2016.

Box 2 - The THALIS-SAGE project

The THALIS-SAGE project (*THALIS - SAGE. Conservation through religion. The sacred groves of Epirus*) was a four-year project financed by the Operational Program Education and Lifelong Learning of the National Strategic Reference Framework. The project, coordinated by the University of Ioannina, aimed at exploring both the history and ecology of a network of sacred forests located in Northwestern Greece. This was the first major and multidisciplinary study investigating Greek sacred forests at this level of detail.

The focus of the project was on three main aspects:

1. Assessing the biodiversity level of Greek sacred forests

The research team studied the biodiversity of eight sacred forests in comparison with corresponding control areas in adjacent forest areas with comparable species composition. The biodiversity assessment involved several species of flora and fauna (e.g. woody plants, passerine birds, bats, lichens, fungi and insects). This study found that sacred forests have a small but persistent biodiversity advantage in comparison to control sites. In particular, we found that there are more distinct sets of organisms amongst sacred groves than amongst control sites (Avtzis et al., 2018).

2. Unravelling the history of sacred forests through dendrochronology techniques

The research team identified and mapped 22 different groves and delimited their boundaries using historical aerial imagines. In all these forests a sample of large trees were aged revealing that overall the forests are 250-350 years old, established somewhere between 1660 and 1760. The research team also prepared environmental educational materials for schools on the ecological and cultural values of trees in sacred forests.

3. Assessing the socio-cultural drivers behind the forests establishment through ethnographic research and archive data

The research team studied the groves history through ethnographic research and historical documents. Their findings shed lights on the different prohibition regimes which were established in the forests (i.e. dedication and excommunication; see Sections II.2.2 and II.2.1).

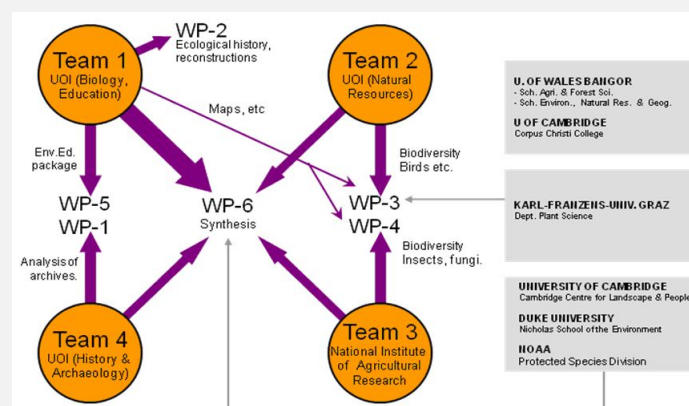


Figure II.2: The THALIS-SAGE project research plan

Table II.1: Cultural, historical and environmental characteristics of the six studied sacred forests.

<i>Cultural and historical information</i>							<i>Forest characteristics</i>				
Sacred forest name	Village name	Type of ritual protection ^a	Forest management practices ^a	Dedicated church	Former population (year)	Current population (Census 2011) ^b	Forest type	Main canopy specie	Size (ha)	Min-max altitude (m) ^c	Main aspect
Livadakia	Vitsa	Dedication, communal agreements	Protection of large trees, grazing, dead wood collection allowed	Church of Agios Nikolaos	940 (ca.1750) ^d	102	Broadleaved oak	<i>Quercus</i> spp. ^f	4.9	881-921	-
Kouri	Mazi	Dedication, communal agreements	Protection of large trees, grazing, dead wood collection allowed	Church of Evangelistria, Church of Proph�tes El�as,	188 (1895) ^e	187	Broadleaved oak	<i>Quercus</i> spp. ^f	10.4	538-648	NE
Mereao	Palioseli	Excommunication, communal agreements	Protection of large trees, occasional grazing and wood cutting	None	1044 (1895) ^e	45	Pine	<i>Pinus nigra</i>	24.4	1107-1319	SW
Agia Paraskevi	Vovoussa	Dedication	Strict protection	Church of Agia Paraskevi	210 (ca.1750) ^d	102	Pine and mixed broadleaf	<i>P. nigra</i>	6.7	966-1059	N
Toufa	Greveniti	Excommunication	Strict protection	None	1500 (ca.1750) ^d	193	Beech	<i>Fagus. sylvatica</i>	117.2	1079-1457	W
Gradista	Kapesovo	Dedication, excommunication, communal agreements	Protection of large trees, grazing, dead wood collection allowed	Church of Agios Nikolaos	400 (ca.1750) ^d	51	Mixed broadleaf	<i>Quercus</i> spp. ^f	23.6	1133-1227	S

^a Classification follows Stara et al. (2016).

^b Population figures are likely to overestimate permanent residents, as many people only live in villages in the summer. Permanent residents can be 30% lower than National Statistics (Labrianidis and Bella 2004).

^c Altitude values have been recomputed from field data for the present study from the values reported by Stara et al. (2016).

^d Source: Labridis (1870).

^e Source: Ottoman Census of 1895.

^f Deciduous oaks: *Quercus frainetto*, *Quercus. cerris*, *Quercus. pubescens*, and (only in Mazi) *Quercus trojana*.

II.2.1 Agropastoralist villages - Sacred forests of Vitsa, Mazi, and Kapesovo

At middle and lower elevations, the vegetation of the area is dominated by a mixed broadleaf forest zone, where deciduous oaks are conspicuous in number. The inhabitants of the oak zone are traditionally made of permanent agriculturalist and transhumant pastoralist communities (Stara et al., 2016). Pastoral woodlands thus dominate the cultural landscape, as fodder is an important subsistence resource for livestock both in winter and summer. The sacred forests of Vitsa, Mazi and Kapesovo are characterized by deciduous oaks and broadleaf trees. They are small sacred forests located close to their village. In the case of Vitsa, the forest used to be set in the proximity of Vitsa ancient settlement, which has been completely abandoned and had disappeared (Stara et al., 2016). Vitsa and Mazi are forests considered sacred because they are dedicated to either a Saint (e.g. *Agios Nikolaos* in Vitsa and Kapesovo) or the Virgin Mary (*Evangelstria* in Mazi). In these forests, the Saints celebration often coincide with important dates, which regulate pastoral activities and open the sites to livestock (Stara et al., 2016). In sacred forests such as these, controlled grazing and browsing by domestic herbivores is allowed, at least for a portion of the year, while the cutting of trees is traditionally prohibited (Fig.II.3). In all but one village the number of inhabitants has drastically dropped during the last centuries (Table II.1).



Figure II.3: Cattle crossing the main gravel road leading to Vitsa sacred forest. ©Marini Govigli V.

II.2.2 Woodcutting communities - Sacred forests of Greveniti, Palioseli, and Vovoussa

In the south-east area of Zagori, the local communities are mainly descendants of Vlach nomads who permanently settled in the area from the 16th century as pastoralists and later being engaged in livestock and forestry activities (Campbell, 1964). Greveniti and Palioseli sacred forests are both large wood belts located above the village, and serve as protection from falling stones and to replenish their water aquifers (Tsiakiris et al., 2019) (Fig. II.4). Both forests were created with an excommunication ritual (*aphorisms*), which threaten anyone trespassing or doing damage to the forest with direct exclusion from the Church (Stara et al., 2016). This regime gives the strongest protection status to these forests. Vovoussa sacred forest is dedicated to the patron saint *Agia Paraskevi* of the church located at the centre of the forest. The forest also has strong strict prohibition rules still in place, which prohibits even hunting or honey collection within its extent (Stara et al., 2016). Similarly to the villages in the agropastoralist zone, depopulation has had a strong impact on the population of all three of these villages. The sacred forests are currently subsumed within the surrounding landscape in a continuous forest layer.



Figure II.4: An example of a rolling stone precariously held by an oak in Palioseli sacred forest. High slope and soil instability often lead to land slide phenomena in the forest. ©Marini Govigli V.

II.3 An interdisciplinary methodology

A diverse set of data stemming from different sources (ecological data, ethnographic data) and of different types (qualitative, quantitative) was collected and employed in this thesis, to examine the nature of the relationship occurring between social and ecological processes in Greek sacred forests. Although each chapter assesses the research objectives with different focuses, assumptions and data sources (Table I.1, Chapter I), we here provide an overview of the main set of ecological and social data employed in the thesis.

II.3.1 Tree data Forest inventory for population parameters

The ecological data on trees collected in the sacred forests was obtained through a forest inventory. Data is principally related to: (i) forest composition (tree species population abundance), (ii) forest structure (tree size in terms of diameter and height), and (iii) forest age (derived from tree core samples). The forest inventory protocol was developed at the beginning of 2014 summer field season and with some modifications was also applied during the 2015 summer field season.

A systematic sampling design based on sample plots located on a square grid was followed for the tree inventory. Notional boundaries of each sacred forest site were defined by Tsiakiris et al., 2013 as the forest extent on ortho-rectified aerial photographs taken in 1945. It was decided that a minimum of 30 plots were needed to characterise the tree populations. Since the forests were of variable sizes, the dimensions of the grid were scaled to the size of each forest. The required grid spacing (S) in meters was estimated using Eq. 1:

$$S = 100 \sqrt{\frac{A_f + A_b}{n}} \quad (1)$$

Here n (the number of grid intersections, i.e. the plot locations) was set at 30 by solving Eq. 2:

$$n = \frac{1}{\left(\frac{E}{t \times CV}\right)^2 + \frac{1}{N}} \quad (2)$$

where E is the allowable error percent (set at 20%), t is the t-value (set at 2), CV the coefficient of variation, and N the population size (after Wong et al., 2007). Information on N and CV were obtained from a pilot survey, which was conducted at the beginning of the 2014 sampling season. The notional area of the forest derived from the 1945 aerial photographs (A_f), was extended to include a 50 m outside buffer (A_b). Plots lying partially outside $A_f + A_b$ were not included (Fig. II.5), resulting in 161 plots in total being recorded across the six sites (Table II.2).

Table II.2: Grid spacing and total number of plots surveyed for the six sacred forests.

Sacred forests	Number of plots surveyed	Grid spacing (m)
Vitsa	29	60
Mazi	29	80
Palioseli	19 ^a	110
Vovoussa	30	45
Greveniti	28	170
Kapesovo	26	60
	161	

^a In Palioseli the maximum possible number of sample plots was restricted to 19.

Each grid intersection defined the south-western corner of three nested plots. Three different plot size were used (Fig. II.6):

- A 15 m × 15 m plot, where all trees with a diameter at breast height (d_1) $\geq$ 5 cm were identified and their d_1 measured;
- A 5 m × 5 m sub-plot, where all saplings and/or shrubs (saplings: diameter at ground layer $d_0 \geq$ 1 cm and $d_1 <$ 5 cm; shrubs: all other woody plants - e.g. showing no leading stem, width greater than height, leaves at ground layer) were identified and their d_0 measured;

- A 2 m × 2 m sub-sub-plot, where all seedlings of woody species (individuals with $d_0 < 1$ cm) were identified³.

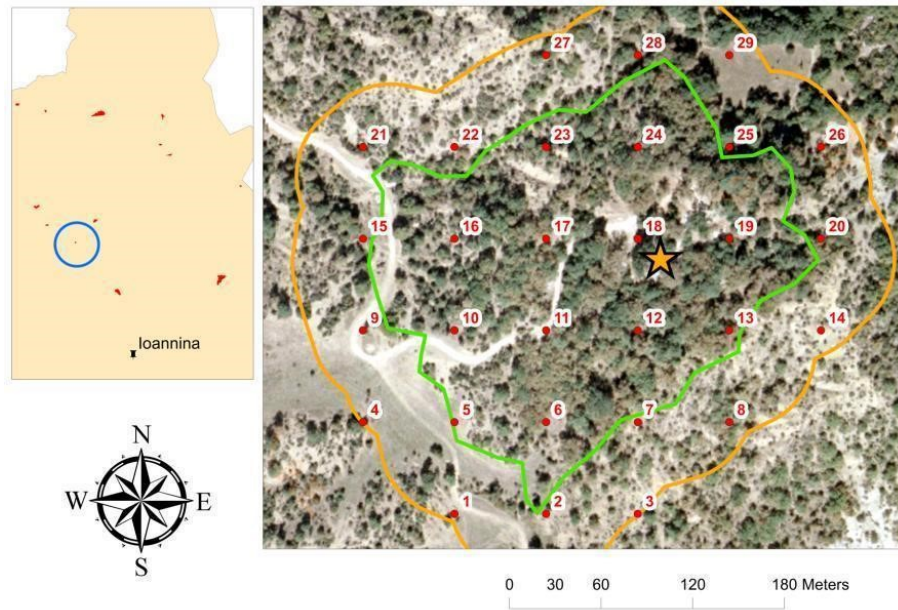


Figure II.5: Map of one study area (Vitsa) as an example of the systematic sampling design undertaken. The map shows the location of survey plots, and the location of the forest within Epirus. Numbers are survey plot identifiers. The green line indicate the provisional boundary of the sacred forest (A_f), while the orange line represents the additional outer 50m buffer zone (A_b). Plots lying between lying partially outside $A_f + A_b$ were not included, as for instance the two survey plots right above plot 21, and 22.

Herbaceous species were excluded from the data collection. Additional variables were recorded for each plot including:

- geographical variables (aspect, slope angle and altitude);
- ground cover and soil properties (leaf litter depth, percentage of vegetation ground cover, deadwood abundance, and soil compaction, depth and stoniness);
- forest management indicators (presence of human litter, presence of human infrastructure in the plot or in close proximity, presence of cut or pruned trees,

³Sub-sub plot were reduced to 1 m × 1 m in the 2015 field season, due to the high number of regeneration materials in sacred forests dominated by black pines and beech.

indicators of forest prohibition regimes including former and active grazing);

- forest structure (top height of canopy, total woody species seedling density, total shrub volume, density of trees and understorey woody plants).

The variables recorded for each of the plots are summarized in Table II.3.

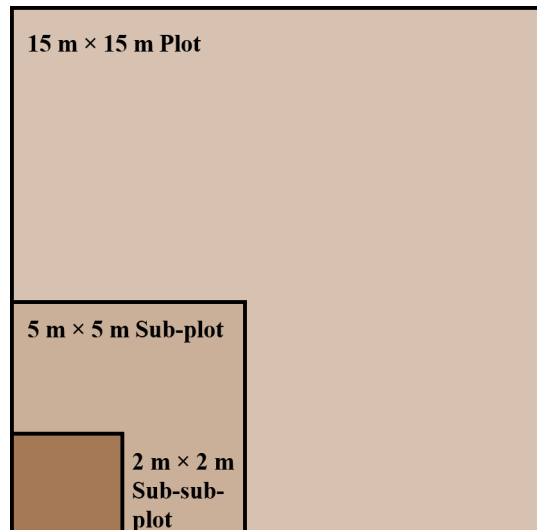


Figure II.6: Visualization of the plot nested structure. During the 2015 field season the sub-sub-plot was restricted to 1 m x 1 m, due to the high number of regeneration materials in sacred forests dominated by black pines and beech.

Table II.3: Variables recorded during the forestry inventory.

Plot level	Variables	Explanation	Measurement
Measured in the 15 m × 15 m plot	Tree census	Identification of the tree species with $d_i \geq 5$ cm	Species ID
	Stem diameter	Tree diameter at 1.3 m (following for particular cases) ²	Measured with a diameter tape, and recorded in cm. Continuous variable.
	Top height of canopy	Height measurement of the highest tree in the plot	Measured with a clinometer in m. Continuous variable.
	Tree density	Density of trees ≥ 5 cm diameter at breast height in the 15 m x15 m plot	Measured as number of trees per plot area (m ²) Continuous variable.
	Northerly aspect	The plot condition of being northerly in aspect	Measured with a compass in degrees (aspect) and converted to Northerly aspect = $\cos(\text{aspect})^1$. Continuous variable.
	Easterly aspect	The plot condition of being easterly in aspect	Measured with a compass in degrees and converted to Easterly aspect = $\sin(\text{aspect})^1$. Continuous variable.
	Slope angle	The slope angle of the plot	Measured with a clinometer in degrees. Continuous variable.
	Altitude	Altitude of the plot	Measured with a GPS in hundred meters. Continuous variable.
	Leaf litter depth	Depth of leaf litter in the plot	Assessed with calipers in cm and then transformed into an ordinal categorical variable: 0 "no leaf litter"; 1 "moderate leaf litter"; 2 "deep leaf litter".
	Vegetation ground cover	% cover of herbaceous plants and woody species versus bare ground in the plot	Assessed descriptively and then transformed into an ordinal categorical variable: 0 "bare ground, no herbs and woody species seedlings"; 1 "sparse herbs and woody species seedlings (max 50% of bare ground)"; 2 "abundant (>50%)" herbs and woody species seedlings in the plot".
	Deadwood abundance	Presence of deadwood in the plot	Assessed descriptively and then transformed into an ordinal categorical variable: 0 "no deadwood"; 1 "sparse deadwood"; 2 "medium deadwood"; 3 "abundant deadwood".
	Wood decay class	Only for dead trees, assessment of the wood decay	Assessed descriptively (from 0 –none to 5 – advanced state of decay)
	Soil compaction	Level of compaction of the soil	Assessed descriptively and then transformed into a binary variable: 0 "loose soil"; 1 "compact soil".
	Soil stoniness	Presence of stones in the soil	Assessed descriptively and then transformed into an ordinal categorical variable: 0 "no stones"; 1 "some stones"; 2 "abundant stones".
	Soil depth	Soil depth	Assessed descriptively and then transformed into an ordinal categorical variable: 0 "shallow soil"; 1 "medium depth soil"; 2 "deep soil".

	Human litter	Presence of human litter (paper, plastics, bullets, etc.) in the plot	Assessed descriptively and then transformed into a binary variable: presence/absence of human litter.
	Harvested or worked trees	Presence of cut or pruned trees in the plot (e.g. branches sawn, cut stumps, pollards, sap-mined trees, etc.)	Assessed descriptively and then transformed into a binary variable: presence/absence of harvested trees.
	Infrastructure	Presence of human infrastructure in the plot and its proximity (e.g. church, icon stands, paths, roads, fences etc.)	Assessed descriptively and then transformed into a binary variable: presence/absence of human infrastructure.
	Prohibition index	Indicator of the prohibition regime of the forest	Measured as 0-3 discrete indicators (0 "full protection, no deadwood extraction"; 1 "controlled management, limited deadwood extraction"; 2 "protection of mature trees only, wood pasture", 3 "protection of mature trees only, wood pasture with traces of current active grazing".
Measured in the 5 m × 5 m sub-plot	Tree census	Identification of the tree species with diameter at ground layer $d_0 \geq 1$ cm and $d_1 < 5$ cm;	Species ID
	Basal diameter	Tree diameter at 0 m	Diameter at ground level zero (cm). Only measured for saplings Measured with a caliper, and recorded in cm. Continuous variable.
	Shrub volume	The crown volume of each shrubs in the 5 m × 5 m sub-plot	Length of the long and short axis (assuming an elliptical horizontal surface of the crown) and height measured in m, Converted to volume in m ³ . Computed for each shrub individually. Continuous variable.
	Understorey height	Height measurement for each sapling and shrub	Measured in 10 cm classes using a measuring stick
	Understorey density	Density of saplings (diameter at ground layer ≥ 1 cm and diameter at breast height < 5 cm)	Measured as the numbers of saplings per sub-plot area (m ²). Continuous variable.
Measured in the 1 m × 1 m sub-sub-plot	Tree census	Identification of the tree species for individuals with $d_0 < 1$ cm)	Species ID (genus level)
	Seedling height	Height measurement for seedlings	Measured in 10 cm classes using a measuring stick
	Total seedling density	The total number of woody species' seedlings per m ² recorded in the 1 m x1 m sub-sub-plot	Measured as total seedlings/sub-sub-plot area (m ²). Continuous variable.

¹ Rodriguez-Moreno, V.M. & Bullock, S.H. (2014). Vegetation response to hydrologic and geomorphic factors in an arid region of the Baja California Peninsula. *Environ. Monit. Assess.*, 186: 1009–1021.

² For specifications see Pion (2014)

II.3.2 Tree data - Wood cores procedures and dating

Wood cores were extracted from trees of the dominant species in each forest at a height between 0.6 and 1 m. Cores were collected during the forest inventory by coring the second largest tree in each plot, so as to avoid potential sampling hazards deriving from large trees being more apt to be hollow or rotten. In addition, a few tree species were found to be unsuitable for age analysis, being: (i) tree species with extremely hard woods (*Quercus coccifera*), (ii) tree species with no clear defined rings (*Acer spp.*, *Carpinus spp.*, and *Ostrya carpinifolia*), (iii) tree species locally infrequent in the sampling areas (*Juniperus spp.*, *Quercus petraea*) (Table II.4). In a second stage of sampling, additional trees were selected for coring within each of the sampled sites to fill in gaps in the diameter size range of cored trees. Only one core was taken per selected tree, to respect the sensitivity of the local communities to tree damaging in their sacred forests.

Coring was conducted following the procedure outlined in Phipps (1985). It is generally recommended that cores are counted fresh but in the low humidity and warm air this proved impossible. Cores were thus mounted dry and prepared by polishing them with progressively finer grades of sandpaper. Rotten or damaged cores and those with rings that could not be distinguished were excluded from the analyses. Cores were then scanned at 1200 dpi, and their annual width increments recorded using the tree ring dating software Coorecorder (Cybis Elektronik & Data, 2013b) and CDendro (Cybis Elektronik & Data, 2013a) (Fig. II.7). The number of corers per species and sites in provided in Chapter IV, Table II.3).

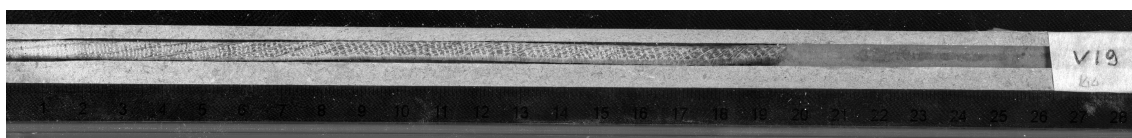


Figure II.7: A scan of a polished oak sample showing the increment rings. The photo was uploaded into Coorecorder software to identify the annual width increments.

Table II.4: Main local tree canopy species and their suitability to tree-ring research. Tree species in bold are the one selected for age analysis.

Tree species	Suitable for coring	Suitable for ring counting	Notes
<i>Acer spp.</i>	Y	N	No clear defined rings
<i>Carpinus spp.</i>	Y	N	No clear defined rings
<i>Fagus sylvatica</i>	Y	Y	
<i>Juniperus spp.</i>	Y	Y	Locally infrequent
<i>Ostrya carpinifolia</i>	Y	N	No clear defined rings
<i>Pinus nigra</i>	Y	Y	
<i>Quercus cerris</i>	Y	Y	
<i>Quercus coccifera</i>	N	-	Extremely hard wood
<i>Quercus frainetto</i>	Y	Y	
<i>Quercus petraea</i>	Y	Y	Locally infrequent
<i>Quercus pubescens</i>	Y	Y	
<i>Quercus trojana</i>	Y	Y	

II.3.3 Social data - secondary sources, interview data, and ethnographic research

A series of social, historical and cultural data on the sacred forests were retrieved to aid the reconstruction of the socio-ecological history of sacred forests in North western Greece. These series of data included: (i) secondary sources and grey literature on the village history, population data, and main management practices; (ii) semi-structured interviews conducted with relevant stakeholders of the sacred forests (villagers, shepherds, local authorities), (iii) key informants knowledge and (iv) field visits with local informers as to record toponyms, boundaries elements, and relevant anecdotes.

In this subsection, a brief overview of such datasets is provided, as to clarify (a) which data was specifically collected via social research methods – i.e. interviews, key informant knowledge, field visits – directly by the PhD researcher for this thesis, (b) which social data was gathered by others, (c) and how such data was integrated and

analysed in the thesis.

Secondary sources and grey literature on the village history, population data, and main management practices

This first set of qualitative data included archive and historical information on the sacred forests and related villages (e.g. village establishment date – if any –, main management practices occurring in the forest, ethnicities, etc.), including economic (e.g. main economic activity of the villagers) and demographic census data at village level. This dataset was compiled using data gathered by others researchers: Dr. K. Stara (e.g. Stara et al., 2016, 2015a,b, but also Stara's unpublished personal archives), A. Efthimiou (Efthimiou, 2019), and Prof. V. Nitsiakos (Nitsiakos, 2016), and integrated in the thesis as follows:

- In Chapter III, this data was used to identify the main prohibition regimes in each of the forest, which was then used to build a prohibition index, indicating the level of disturbance existing in the forests (from full protection to prescribed grazing). Moreover it was finally used to relate the vegetation structure of the forests to different management regimes.
- In Chapter IV, this data – mainly the population data at village level– was used to trace back different waves of tree recruitment in five sacred forests to demographic shifts.
- In Chapter VI, population data and management changes were used to develop social scenario to assess the effect of social changes over the spatial extent of a simulated sacred forest.

Semi-structured interviews and ethnographic research conducted with relevant stakeholders of the sacred forests

A series of semi-structured interview were conducted by the PhD researcher, with the support of Dr. K. Stara and N. Pion, as to identify grazing and management practices

in sacred forests (reported in section V.2.3, and Pion, 2014). Moreover, in this thesis we relied strongly on former ethnographic research undertaken by Dr. K. Stara in the sacred forests (co-author of the relative chapters), and A. Efthimiou (in the forest of Greveniti)⁴. This set of qualitative data was integrated in the thesis as follows:

- In Chapter IV, to interpret and validate qualitatively the findings.
- In Chapter V this information was used to map the main perceptive elements associated with the forest by local communities, as well as to identify the main boundaries elements across different sacred forests.
- In Chapter VI, this data was used to derive the strength of prohibition rules occurring in one sacred forest, which was then integrated in the modelling interface to assess the effect of social changes over the spatial extent of a simulated sacred forest (Scenario 3; VI.3.2).

Field visits with local informers as to record toponyms, boundaries elements, and relevant anecdotes

Field visits were performed with experts and local inhabitants (e.g. as reported in Tsiakiris et al., 2019) in several sacred forests. Similarly to data source number 2, in this thesis we relied also to information retrieved from past field visits with local informants undertaken by Dr. K. Stara, as well as by A. Efthimiou. This set of data was integrated in the different chapters as presented above, and especially proved useful to identify toponyms and mapping cultural boundary elements in Chapter V.

Table I.1 in Chapter I provides further details on how social data was integrated in each of the analytical chapters of the thesis.

⁴All the qualitative data was collected in Greek. For this reason, I am strongly indebted to the direct support of Dr. K. Stara and A. Efthimiou who made available their personal databases of interviews with relevant community stakeholders.

II.3.4 Geophysical and environmental data

Climate data on the study areas were kindly provided by the Metzovo Forestry Department, while geological and hydrological data were obtained from the Institute of Geological and Mineral Research in Preveza (e.g. Fig. II.8). Altitudinal data were derived from GPS units whilst onsite.

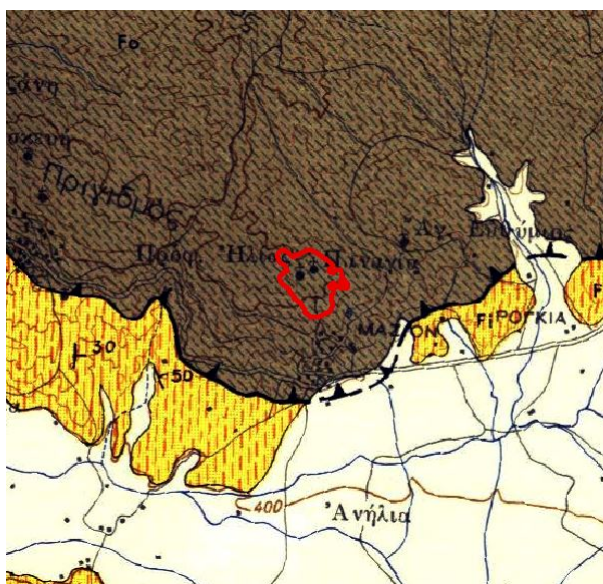


Figure II.8: Example of a geology map. Colours refer to different geological sub-stratum. The area highlighted in red is one of the study areas.

II.3.5 Quantitative and qualitative analyses

The different set of data gathered were integrated and analysed in the thesis using both quantitative and qualitative techniques. Table I.1 in Chapter I displays the main quantitative and qualitative techniques used, which are presented in details in the respective section in which they are deployed.

CHAPTER III

When nature meets the divine: effect of prohibition regimes on the structure and tree-species composition of sacred forests in Northern Greece

III.1 Introduction¹

Sacred forests in northern Greece are an essential component of its mountain cultural landscape, hypothesized to be the result of both ecological processes and site-specific management practices set up by strict religious prohibitions (Stara et al., 2016). These practices acted as constraints on natural forest development by suppressing understorey growth, while prohibitions on woodcutting have preserved large-sized trees. Due to macro-scale socio-economic changes, sacred forests social function and associated management regimes are now fading, with consequent ecological changes in the understorey composition. These changes are expected to be especially significant in the forests where grazing was once permitted. From available oral narratives, historical archive data, and expert field visits, there is evidence that northern Greece sacred forests:

- might had a much more open canopy in the past (as evidenced by the presence of trees with a growth form indicating that they were previously open-grown), due to their unique management regimes (Rackham, 2014);
- might have previously been subject to more active management (as evidenced, e.g. by the presence of relict pollarded trees and former grazing areas; Pion, 2014).

Whilst recent research has revealed that sacred forests in Epirus differ greatly from one another in terms of size, forest type and management practices (Stara et al., 2016; Tsiakiris et al., 2013), the structural and species composition of Greek sacred forests has not been extensively studied. The only available contribution focuses on broad site-level vegetation characteristics (Korakis et al., 2014). This is a research gap worthwhile investigating, as disentangling the forests tree species composition and exploring how

¹The analysis and results of this chapter are published in *Web Ecology*, an international peer-reviewed Journal (Marini Govigli et al., 2020). Moreover, they have been presented in two scientific conferences (8th HELECOS Ecology, Thessaloniki, Greece, 2016 poster presentation; 18th Congress of Hellenic Forestry Society, Edessa, Greece, 2017: peer-reviewed conference proceedings; Marini Govigli et al., 2017)

species react to different management practices is crucial for understanding how sacred forests' ecology change in different management scenarios (e.g. active management vs. abandonment). Additionally, given that such forests play a role in conserving biodiversity, it is important to know how much of the forests social function remains active after abandonment and for how long.

In this study, we investigate the interaction of environment and management regimes on the structure and tree species composition of sacred forests. The main goal is determining whether in sacred mountain forests there are dissimilar types of woody plant communities that reflect different prohibition regimes established in the forests. In this context, our objective are to explore: 1) what kind of woody communities are mainly dominating Sacred Natural Sites taking as an representative sample six different montane sacred forests in Epirus, 2) what kind of differences are existing in the trees composition between their canopy and their understorey level, and 3) whether these differences are linked to abandonment, specific management regimes or physical environment.

We do this by (i) assessing differences in composition between understorey and canopy vegetation, (ii) identifying how woody plant communities are differentiated by both environmental-driven gradients and former versus current management regimes of the forests. Our results will allow us to both identify common characteristics in tree species composition across different forests and structural stages and link them to environmental and social variables potentially related to local management regimes. Understanding the structural and tree-species composition of Greek sacred forests is especially important when the landscape faces an abandonment scenario, as has occurred widely across most of the Northern Mediterranean rural areas.

III.2 Material and methods

Six sacred forests were selected for the present study, as presented in Chapter II. These were: (i) *Agios Nikolaos* in Livadakia, belonging to the village of Vitsa (ii) *Kouri*, village of Mazi; (iii) *Mereao*, village of Palioseli; (iv) *Agia Paraskevi*, village of Vovoussa; (v) *Toufa*, village of Greveniti, (vi) *Graditsa*, village of Kapesovo. Additional information on the forests is presented in Chapter II.

III.2.1 Data collection

During the summers of 2014 and 2015, a tree inventory was undertaken in each site. A systematic sampling design was used by fitting a square grid orientated to cardinal directions over the site area. Three different plot size were used:

- A 15 m $\times$ 15 m plot, where all trees with a diameter at breast height (d_1) $\geq$ 5 cm were identified and their d_1 measured;
- A 5 m $\times$ 5 m sub-plot, where all saplings and/or shrubs (saplings: diameter at ground layer $d_0 \geq$ 1 cm and $d_1 <$ 5 cm; shrubs: all other woody plants - e.g. showing no leading stem, width greater than height, leaves at ground layer) were identified and their d_0 measured;
- A 1 m $\times$ 1 m sub-sub-plot, where all seedlings of woody species (individuals with $d_0 <$ 1 cm) were identified.

The detailed description of the inventory as well as all forest and management variables mapped is provided in Chapter II. The main variables used for this study included:

- geographical variables (aspect, slope angle and altitude);
- ground cover and soil properties (leaf litter depth, percentage of vegetation ground cover, deadwood abundance, and soil compaction, depth and stoniness);
- forest management indicators: presence of human litter, (e.g. gun cartridges) presence of human infrastructure (e.g. buildings, icon stands, paths, road) in the

plot or in close proximity, presence of cut or pruned trees, indicators of forest prohibition regimes including former and active grazing (e.g. recent browsing damage, animal footprints, presence of grazing animals, historical information on the forest management status);

- forest structure (top height of canopy, total woody species seedling density, total shrub volume, density of trees and understorey woody plants).

III.2.2 Data analysis

To investigate the tree diversity of each site, we constructed species rank abundance distributions (RADs) for both canopy (15 m $\times$ 15 m plots) and understorey (5 m $\times$ 5 m sub-plots) layers. In this analysis, we used only the tree species found in the canopy. Thus all shrub and herbaceous species were excluded. Additionally, the deciduous oaks (*Quercus cerris*, *Q. frainetto*, *Q. petraea*, *Q. pubescens*) were lumped at the genus level (*Quercus spp.*) for the sub-plot data. This was done because of the difficulty of identifying individuals in the sapling and shrub size classes to the species level within this group. Plot-level data of deciduous oaks were also pooled at the genus level as this analysis aimed to compare understorey and canopy data. RAD curves provide valuable information about species evenness (the steepness of the curve) and richness (the number of species sampled) (Magurran, 2004). The frequency of sampled tree species was used to examine variation in woody species composition between canopy and understorey layers of the sample sites via a Wilcoxon signed-rank test. This test allowed us to assess whether their population mean ranks differed.

The purpose of this analysis is to categorize the woody vegetation of the six sacred forests according to their vegetation composition and structure, and then assess the respective influence of physical, environmental, and forest management variables. Although sacred forests have been previously classified into two distinct groups based on livelihoods and prohibition regimes (dedication vs. excommunication; Stara et al. (2016); agropastoralist sites vs. woodcutting communities; Sections II.2.1 and II.2.2),

none of these classifications took as starting point the vegetation compositions of these sites both at canopy and understorey levels.

The categorization of the woody vegetation follows a four-stage methodology. Firstly, we computed the Bray-Curtis measure of dissimilarity to assess tree composition difference amongst plots across sites using Bray-Curtis-based non-metric multidimensional scaling (NMDS). Secondly, we created clusters based on closeness within the relative distance matrix, at both canopy and understorey layers. Thirdly, we merged the two set of clusters in a final configuration of four canopy-understorey groups which were represented across sites. Lastly, we ran a regression model on the woody plant groups with environmental and forest management variables, to explore any significant relationships.

Tree basal area as a measure of dominance per species (B), rather than presence-absence or density, was used to assess dissimilarity in woody composition at canopy layer. Equation 1 gives the standard Basal Area index used, expressed in m^2 :

$$B = \frac{\pi}{4 \times 10^4} \sum_{j=1}^{T_s} d_1(j)^2 \quad (1)$$

where T_s is the total number of trees T per species s and $d_1(j)$ is the diameter of tree- j at breast-height. The analysis of saplings and shrubs in the $5m \times 5m$ sub-plots was undertaken with presence-absence data, due to the more homogeneous size distribution of individuals between $d_0 \geq 1$ cm and $d_1 < 5$ cm. The plots that had no tree or shrub cover in the sampled size classes at either plot or sub-plot layers (18 overall) were removed from the analysis of woody species composition.

The resulting distance matrix was then repeatedly clustered (iteratively from 2 to 10 clusters) using a k-means clustering protocol with 25 random starts (Aerts et al., 2006; Dufrêne and Legendre, 1997). A measure of cluster homogeneity (silhouette) was used to identify the most explanatory cluster level. A configuration of five clusters was selected as a meaningful partition of the sample at the canopy/plot layer. Four clusters were selected for the understory/sub-plot layer. Indicator species analysis was applied

to determine which group of tree species best characterised each of the selected clusters. The indicator species analysis was run following the prescriptions of Dufrêne and Legendre (1997) and applying 1000 permutations. Similarly, only those indicator species that emerged as significant ($p < 0.05$) and with an indicator value $> 25\%$ were considered (De Cáceres et al., 2012).

In order to investigate the association of environmental gradients with variation between the plot/sub-plot clusters, non-metric multidimensional scaling (NMDS) was run on the clusters based on plot basal area measurements (and presence-absence data for the sub-plots)². NMDS was performed with three dimensions, a Bray-Curtis measure of distance, and a maximum number of 500 iterations. The relationship between the resulting NMDS axes and each plot environmental variable was tested with multiple linear mixed models. Each model included a fixed effect of site, controlling for the nested design of the sampling strategy (i.e. a series of plots sampled across six sites). Additional analyses were run to test the multiple regression assumptions for all models (linearity, homoscedasticity, independence and normality). Due to the spatial nature of the study dataset (systematic grids within six study sites), Moran's autocorrelation coefficient was used to quantify whether NMDS outcomes were affected by the presence of spatial autocorrelation (Paradis, 2019). The test was performed on the three NMDS axes, using as distance weight matrix the plots coordinates (Greek grid). The results of the test showed a significant positive autocorrelation across observations for all resulting NMDS axes at both plot and sub-plot levels. The spatial correlation trend existing in the data was inspected using variogram plots. This assessment identified a linear correlation structure as the best model for the spatial correlation of the data. This variogram model was then fitted to all regression models (except one model at sub-plot level, which was better fitted using an exponential spatial correlation structure) in order

² As an alternative analysis, we performed a Canonical Correspondence Analysis (CCA) on the tree similarity matrix and related environmental variables (Appendix A). We found that the results yielded by this analysis are comparable to the NMDS.

to filter out the effect of the spatial correlation on the analyses. Explanatory variables were tested for multi-collinearity, using a Generalized Variance Inflation Factor (GVIF; Fox and Monette, 1992). Variables showing a GVIF greater than 2 were removed. Subsequently, a backward stepwise regression analysis based on the model Akaike Information Criterion (AIC) was used to select the most parsimonious model (Akaike, 1974). In order to achieve normality of residuals, Box-Cox transformations were applied as required. Only in one case (the first NMDS axis of the sub-plot cluster analysis) did the regression assumptions test fail, due to a non-linear distribution of model residuals. A simple measure of correlation (Kendall rank correlation coefficient) between environmental variables and the corresponding NMDS axis was used in that case.

Kruskal-Wallis non-parametric tests of differences were performed to examine whether environmental variables were distributed differently across clusters at plot and sub-plot levels. This analysis helped to understand how to cross-reference plot and sub-plot data into new groups in order to assess the pattern across the size classes of woody plants simultaneously. Post-hoc comparisons were run to confirm if differences occurred between groups, at mean and median levels of environmental and forest structure variables. Because of ties in data (observations having the same value), the Nemenyi-test with Chi-squared approximation was used (Pohlert, 2014). The relationship between the four groups emerging from the combination of canopy/plot and understorey/sub-plot and correlated environmental variables and forest management indicators was explored using an ordered logit regression and quantified using relative odds ratios. The logit model was fitted using a mixed effects ordinal logistic regression to account for the nested design of the sampling. Latitudinal data of the plots was employed as additional covariate to accommodate spatial correlation of data (Hu and Lo, 2007). This final analysis assumes that the dependent variable (the four woody plant groups) behaves as an ordered attribute, according to varying probabilities associated with a pool of independent variables. The results of this regression model categorize the different woody plant groups according to variation in

environmental and prohibition regimes. Statistical analysis was performed using the R computing environment (R Core Team, 2017).

III.3 Results

III.3.1 Woody plant communities in the sacred forests and intra-site floristic diversity

The inventory recorded 51 woody species across the six sites, of which 63% present both as canopy trees (in the plot layer) and as understorey saplings or shrubs (in the sub-plots layer), 20% appeared only in the understorey, 9% only in the canopy and 8% only as seedlings (sub-sub-plot layer). Eighteen plots had no tree canopy or understorey and only contained herbaceous plants. The tree canopy of each site is dominated by a handful of species though with some parallels between sites: the canopies in Vitsa, Mazi and Kapesovo are characterized by deciduous oaks (*Quercus frainetto*, *Q. cerris* and *Q. pubescens* and, only in Mazi also including the semi-deciduous *Q. trojana*), Greveniti is dominated by beech (*Fagus sylvatica*), while Vovoussa and Palioseli are dominated by black pine (*Pinus nigra*).

The composition of saplings and shrubs in the understorey is more variable. In Vitsa, Mazi, and Kapesovo a shrubby and sclerophyllous evergreen understorey (dominated by *Quercus coccifera* and *Juniperus oxycedrus*) is present in combination with deciduous oak and oriental hornbeam (*Carpinus orientalis*) saplings, in Mazi and Kapesovo, together with saplings of mock privet (*Phillyrea latifolia*) in Mazi, and saplings of Montpellier maple (*Acer monspessulanum*) in Kapesovo. Palioseli and Vovoussa are characterized by an understorey comprising saplings of hop-hornbeam (*Ostrya carpinifolia*), hornbeam (*Carpinus betulus*) and black pine (*P. nigra*). In Greveniti the understorey is mainly dominated by saplings of beech (*F. sylvatica*).

Species rank-abundance curves for the most abundant tree species in each of the six sites confirmed that overall, the composition and community structure of the woody plants in the understorey broadly matches that of trees in the canopy (Fig. III.1). Wilcoxon signed-

rank tests revealed no significant statistical differences between understorey and canopy compositions. However, there are a few differences worth noting. In Greveniti three species (*Acer opalus*, *P. nigra*, *Fraxinus angustifolia*) and deciduous *Quercus spp.* present in the canopy, were totally missing from the understory. In two sites, some tree species occur at a smaller relative abundance in the understory compared with the canopy layer: *C. orientalis* in Vitsa and deciduous *Quercus spp.* and semideciduous *Q. trojana* in Mazi. In the other three sites, several species were more abundant at understory layer than in the canopy: mainly *C. orientalis*, *Cornus mas*, *Juniperus oxycedrus*, *J. communis* and *Prunus avium* in Vovoussa, mainly *Fraxinus ornus* and *C. mas* in Mazi, and mainly *J. oxycedrus* in Palioseli.

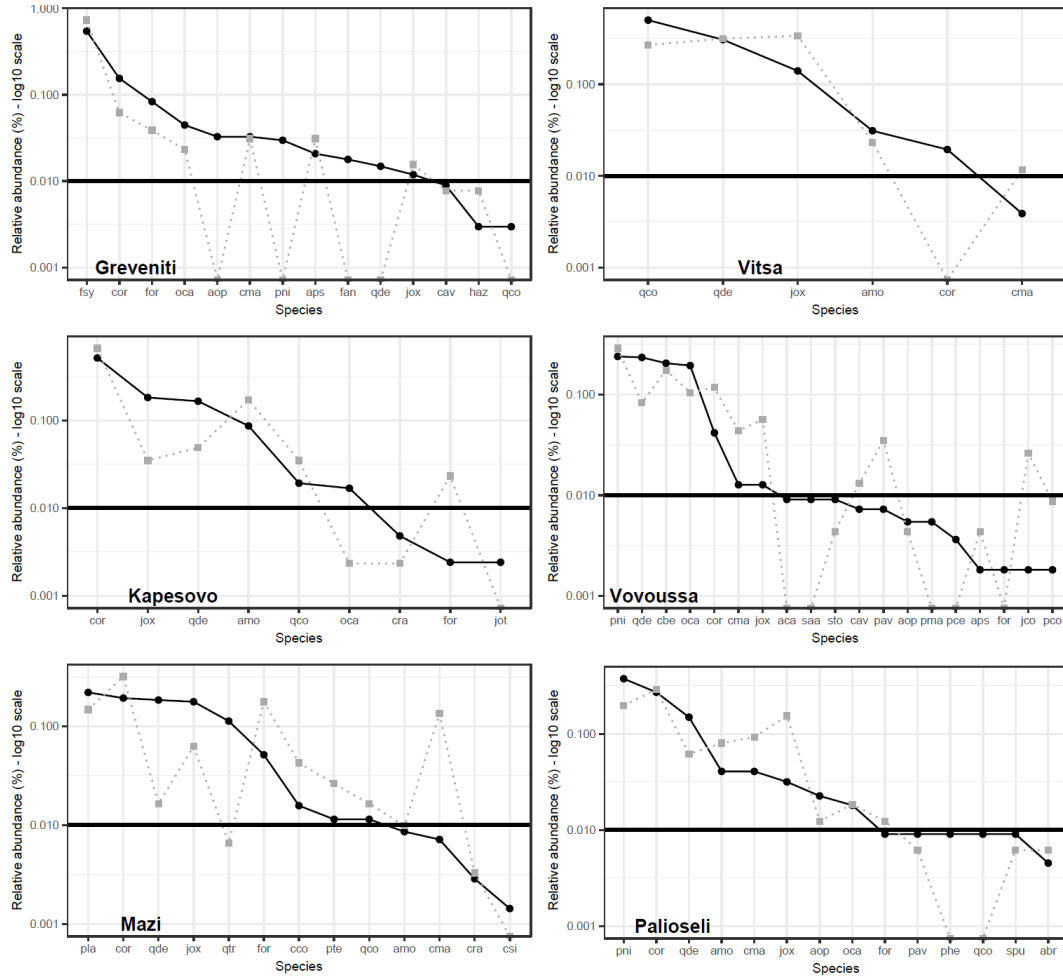


Figure III.1: Tree species rank-abundance curves in the six sacred forests. Black curves indicate the relative abundance of the most common species in the canopy layer (main plots). Gray dotted curves indicate the relative abundance of saplings in the understory layer (sub-plots) matching the tree species found in the canopy. The horizontal straight line sets the abundance threshold of 1%. Full species names are given in Appendix H. Deciduous *Quercus* spp. (*Quercus cerris*, *Q. frainetto*, *Q. petraea* and *Q. pubescens*) are lumped into a single taxon.

III.3.2 Inter-site variation and effects of environmental and management variables

The NMDS ordination of tree species recorded in the plot layer showed three dimensions that produced the greatest reduction in average stress (Aerts et al., 2006). These dimensions were then examined in relation to the three ordination axes to allow some interpretation of the characteristics of the five clusters identified in the plot layer. The first ordination axis partitions northern-facing plots (coefficient -0.24; $t = -3.79$; $p < 0.001$) with greater slope angles (coefficient -0.01; $t = -2.38$; $p = 0.019$) from the others. It also identifies plots associated with the presence of human infrastructure (coefficient 0.18; $t = 2.10$; $p = 0.038$) and higher levels of disturbance associated with forest prohibition regimes (increased wood removal and grazing) (coefficient 0.28; $t = 2.23$; $p = 0.028$). Examination of the species in these plots shows that this first axis partitions cluster 1 dominated by black pine in the canopy and cluster 4 with beech canopy from the other clusters (2, 5) which have canopies of other species (Fig. III.2; further details in Appendix B (Fig.B1), and Appendix C). The second axis partitions clusters 4 and 5 from the rest, and correspond to forests with limited presence of cut or pruned trees (coefficient -0.15; $t = -1.86$; $p < 0.050$). The third axis partitions plots with less compact soil (coefficient -0.13; $t = -1.94$; $p = 0.054$) and separates cluster 3 from the rest.

For saplings and shrubs recorded in the sub-plot layer, the NMDS analysis also shows three dimensions and yields four clusters. Ordination of the first two NMDS axes shows that the first axis separates cluster 1 (dominated by beech) and cluster 4 (mixed broadleaves of *C. betulus* and *O. carpinifolia*) from clusters 2 and 3 (Appendix B (Fig.B2), and Appendix C). Cluster 1 is strongly, and cluster 4 more weakly, negatively associated with stronger forest prohibition regimes that limit disturbance (less wood removal and grazing) (Kendall rank correlation coefficient 0.44; $p < 0.001$), whereas they are positively associated with greater slope angle (-0.32; $p < 0.001$) and altitude (-0.22; $p < 0.001$) amongst other variables. Additionally, the first axis is positively correlated

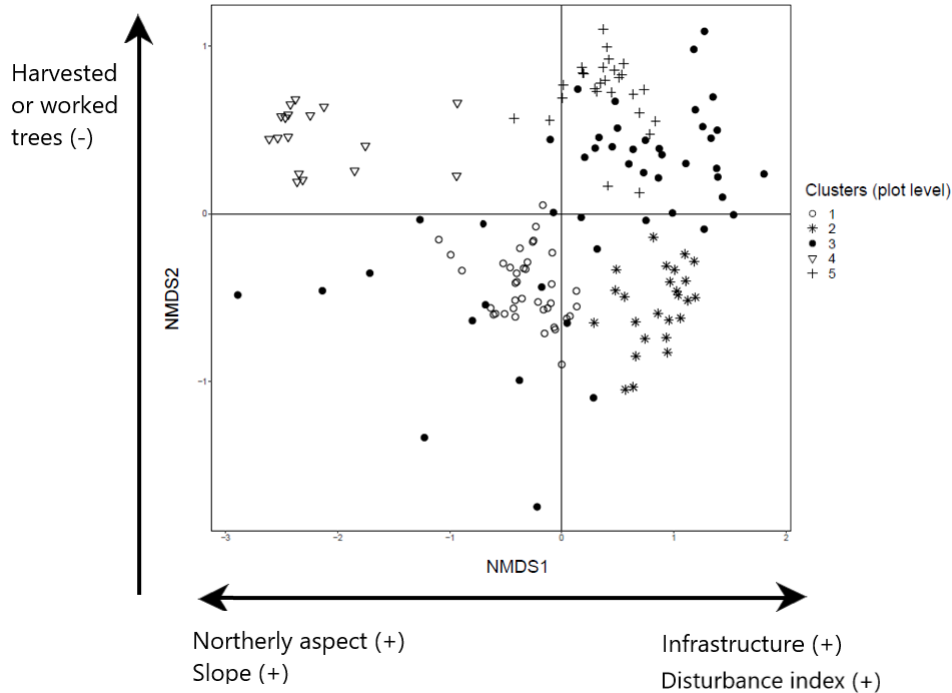


Figure III.2: NMDS ordination (axes 1 and 2) of 154 plots across six sacred forests for canopy trees. Plots are labelled according to the five clusters produced by the cluster analysis (1= Pine (*Pinus nigra*), 2= Mixed oaks (dominated by *Quercus coccifera* and *Q. frainetto*), 3= Sclerophyllous species (*Acer monspessulanum*, *Q. coccifera* and *Juniperus oxycedrus*), 4=Beech (*Fagus sylvatica*), 5= Mixed broadleaves (*Q. pubescens* and *Carpinus orientalis*). Arrows indicate the main forest prohibition regimes and environmental variables statistically related to the axis coordinates. An increase in the disturbance index points to forest prohibition regimes allowing higher levels of disturbance (increased wood removal and grazing).

with low leaf litter depth (0.22; $p = 0.001$), limited deadwood abundance (0.32; $p < 0.001$), and shallow and compacted soil (respectively 0.29; $p < 0.001$, and 0.37; $p < 0.001$). Axis 3, on the other hand, shows a positive and significant association with forest prohibition regimes with higher levels of disturbance (increased wood removal and grazing) (coefficient 0.05; $t = 2.57$; $p = 0.011$), while a negative association with leaf litter depth (-0.07; $t = -2.48$; $p = 0.014$) (Fig. III.3) and separates cluster 2 (dominated by *C. orientalis* and *A. monspessulanum*) from cluster 3.

Significant differences were found for both trees (plot layer) and saplings/shrubs (sub-plot layers) when comparing environmental variables amongst clusters. These

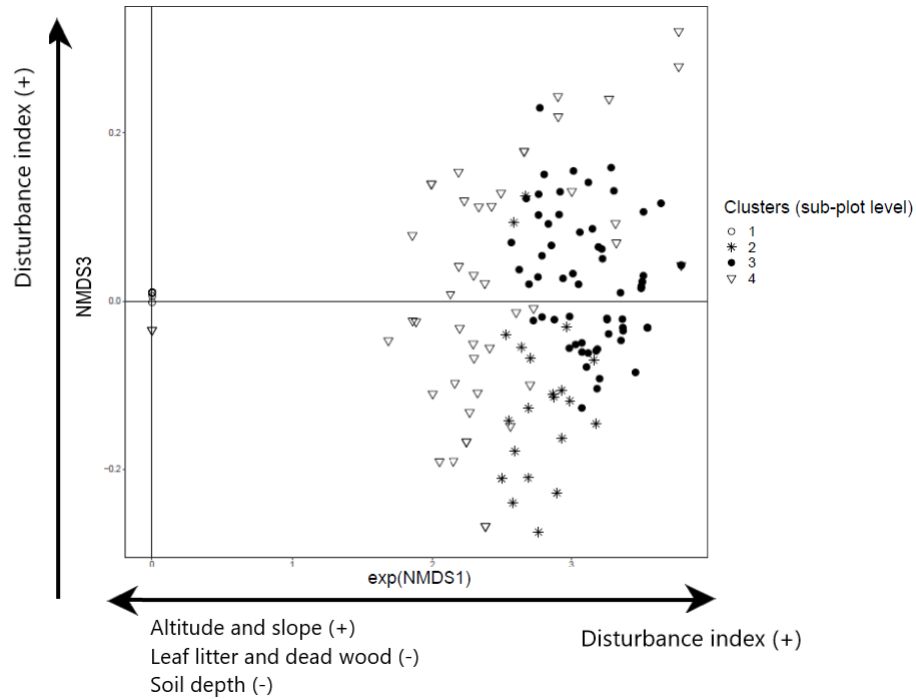


Figure III.3: NMDS ordination (axes 1 and 3) of 147 sub-plots across six sacred forests for understorey saplings and shrubs. Sub-plots are labelled according to the four clusters produced by the cluster analysis (1=*Fagus sylvatica*, 2=*Carpinus orientalis*/*Acer monspessulanum*, 3=No significant species, 4=*Carpinus betulus*/*Ostrya carpinifolia*). The first axis of the NMDS ordination was transformed as $\exp(\text{NMDS})$ to improve readability. Arrows indicate the main forest prohibition regimes and environmental variables statistically related to the axis coordinates. An increase in the disturbance index points to forest prohibition regimes allowing higher levels of disturbance (increased wood removal and grazing).

differences are mostly significant between (i) cluster 1 and 4 and (ii) clusters 2, 3 and 5 for trees (Appendix D), and between (i) clusters 1 and 4 and (ii) clusters 2 and 3 for saplings/shrubs (Appendix E) The results of the two cluster analyses were merged into a final reconfiguration of four woody plant groups, to reflect the pattern across the size classes of woody plants simultaneously (Appendix F) .

The four woody plant groups are:

1. High-canopy forest (average canopy top height 31.6 ± 1.7 m) with an understorey dominated by saplings of single or mixed tree species (HIGH-SAP). This group is dominated by *F. sylvatica* or *P. nigra* - *Q. cerris*. The dense canopy layer shades

the understory dominated by either pole-sized trees of *F. sylvatica*, or saplings of *P. nigra*, *C. betulus* and *O. carpinifolia*, with only a low abundance of shrubs. These forests are characterized by soil with a deep leaf litter layer, and a high abundance of deadwood.

2. High-canopy forest (average canopy top height 24.6 ± 1.8 m) with an understory dominated by short and often shrubby saplings (HIGH-SHR). This group is dominated by *P. nigra* and largely similar to group 1 with the main difference being the denser shrubby nature of its understorey, dominated by species such as *C. orientalis* and *J. oxycedrus*.
3. Low-canopy forest (average canopy top height 15.1 ± 1.8 m) with an understorey dominated by saplings of mixed tree species (LOW-SAP). This group is characterized by oak and mixed broadleaf forest communities (including *Q. cerris*, *Q. pubescens* and *Q. trojana*). These forests had smaller tree diameters and lower tree densities than groups 1 and 2. The understorey is dominated by tree saplings, most commonly of *Cornus mas*, *Fraxinus ornus*, and *P. latifolia*. The soil is characteristically much more compact, shallower and with a thinner litter layer than in groups 1 and 2.
4. Low-canopy forest (average canopy top height 9.8 ± 0.6 m) with an understory dominated by shrubs (LOW-SHR). It has a canopy characterized by small-stature oaks (*Q. pubescens*, *Q. cerris*, *Q. frainetto*), interspersed with patches dominated by dense shrubs. Sclerophyllous vegetation is abundant with the main species being *C. orientalis*, *Q. coccifera* and *J. oxycedrus*. This group has lower abundance of deadwood and a soil with a shallow leaf litter layer compared with the other groups.

The four identified woody plant groups can be assigned to existing classifications of habitat and vegetation types in Europe. According to the EUNIS forest habitat classification (Schaminée and S., 2013), the HIGH-SAP group includes both G1.6 (*Fagus*

woodland) and G3.5 (*Pinus nigra* woodland) habitat types, HIGH-SHR includes G3.5 (*Pinus nigra* woodland), LOW-SAP relates to G1.7 (*Thermophilous deciduous woodland*), while LOW-SHR relates to both G1.7 and G2.1 (*Mediterranean evergreen [Quercus] woodland*). More precisely, LOW-SAP and LOW-SHR can be respectively assigned to the associations *Phillyreo latifoliae-Carpinetum orientalis* and *Verbasco glabrati-Quercetum frainetto* as identified in Bergmeier and Dimopoulos (2008); while HIGH-SHR can be assigned to the vegetation type *Pyrolo chloranthae-Pinetum nigrae* (Bergmeier, 2002). Table III.1 presents the tree composition and abundance for each of the woody plant groups.

When compared at mean and median levels, the four identified woody plant groups constitute ordered categories of sampled environmental and stand variables (Fig. III.4, Appendix G). The low canopy-shrub forest group (LOW-SHR) exhibits the highest disturbance index i.e. (increased deadwood removal and grazing), and the highest shrub volume in the understorey. In contrast, the high canopy sapling forest group (HIGH-SAP) has the highest average canopy height, and the steepest slope. The other groups show intermediate values. This statistically significant ordination across the four groups is also supported by the results of the ordered logit regression (Table III.2; Fig. III.5) with the Odds Ratio showing that, holding all other variables constant, the odds of a plot being a high canopy-sapling forest, rather than a low canopy-shrub forest are:

- increased 305% by an increase in orientation of the aspect of the plot to the north;
- decreased 90% with a shift to a forest prohibition regime with higher levels of disturbance (increased wood removal and grazing);
- decreased 68% by the presence of human infrastructure.

The distribution of the four woody groups across the six forests reveals that the spatial distribution of plots classified into each of the final groups does not map to individual sites but are rather spread across sites, with HIGH-SAP being the most abundant group in Greveniti and Vovoussa, and LOW-SHR the most abundant group in Vitsa, Mazi and Vovoussa (Appendix F).

Table III.1: Species composition and abundance for the four main woody plant groups of dominant canopy trees and understorey saplings and shrubs defined. Only species contributing to a cumulative 90% of the group composition are shown.

Plot inventory (dominant trees)		Subplot inventory (understorey)	
Group HIGH-SAP			
Species	Basal Area %	Species	Number of individuals %
<i>Pinus nigra</i>	59.5	<i>F. sylvatica</i>	26.8
<i>Fagus sylvatica</i>	26.3	<i>P. nigra</i>	25.2
		<i>Carpinus betulus</i>	13.4
		<i>Ostrya carpinifolia</i>	8.9
		Deciduous oaks	5.1
		<i>Cornus mas</i>	3.5
		<i>Juniperus communis</i>	2.6
		<i>Acer monspessulanum</i>	1.6
		<i>J. oxycedrus</i>	1.3
		<i>Prunus avium</i>	1.3
Group HIGH-SHR			
Species	Basal Area %	Species	Number of individuals %
<i>P. nigra</i>	85.9	<i>Carpinus orientalis</i>	36.2
<i>Quercus cerris</i>	7.2	<i>J. oxycedrus</i>	27.6
		<i>P. nigra</i>	15.5
		Deciduous oaks	7.8
		<i>C. mas</i>	6.0
Group LOW-SAP			
Species	Basal Area %	Species	Number of individuals %
<i>Q. cerris</i>	20.3	<i>C. mas</i>	21.8
<i>Q. pubescens</i>	15.7	<i>F. ornus</i>	14.4
<i>Q. trojana</i>	13.4	<i>P. latifolia</i>	10.6
<i>O. carpinifolia</i>	8.9	Deciduous oaks	5.6
<i>C. orientalis</i>	7.4	<i>F. sylvatica</i>	5.9
<i>A. opalus</i>	7.1	<i>O. carpinifolia</i>	5.9
<i>C. betulus</i>	4.2	<i>Cotinus coggygria</i>	5.3
<i>Q. frainetto</i>	3.2	<i>C. orientalis</i>	3.7
<i>J. oxycedrus</i>	3.2	<i>Prunus spinosa</i>	3.2
<i>P. nigra</i>	3.1	<i>Corylus avellana</i>	3.2
<i>Phillyrea latifolia</i>	2.4	<i>Coronilla emeroides</i>	3.2
<i>Fraxinus ornus</i>	2.2	<i>P. avium</i>	2.7
		<i>A. pseudoplatanum</i>	2.1
Group LOW-SHR			
Species	Basal Area %	Species	Number of individuals %
<i>Q. pubescens</i>	27.2	<i>C. orientalis</i>	52.1
<i>Q. frainetto</i>	14.2	<i>J. oxycedrus</i>	11.7
<i>Q. cerris</i>	11.7	<i>Q. coccifera</i>	9.4
<i>C. orientalis</i>	10.5	<i>P. latifolia</i>	6.6
<i>Q. coccifera</i>	6.6	<i>A. monspessulanum</i>	6.5
<i>Q. trojana</i>	6.4		
<i>J. oxycedrus</i>	5.9		
<i>A. monspessulanum</i>	5.2		
<i>P. latifolia</i>	3.0		

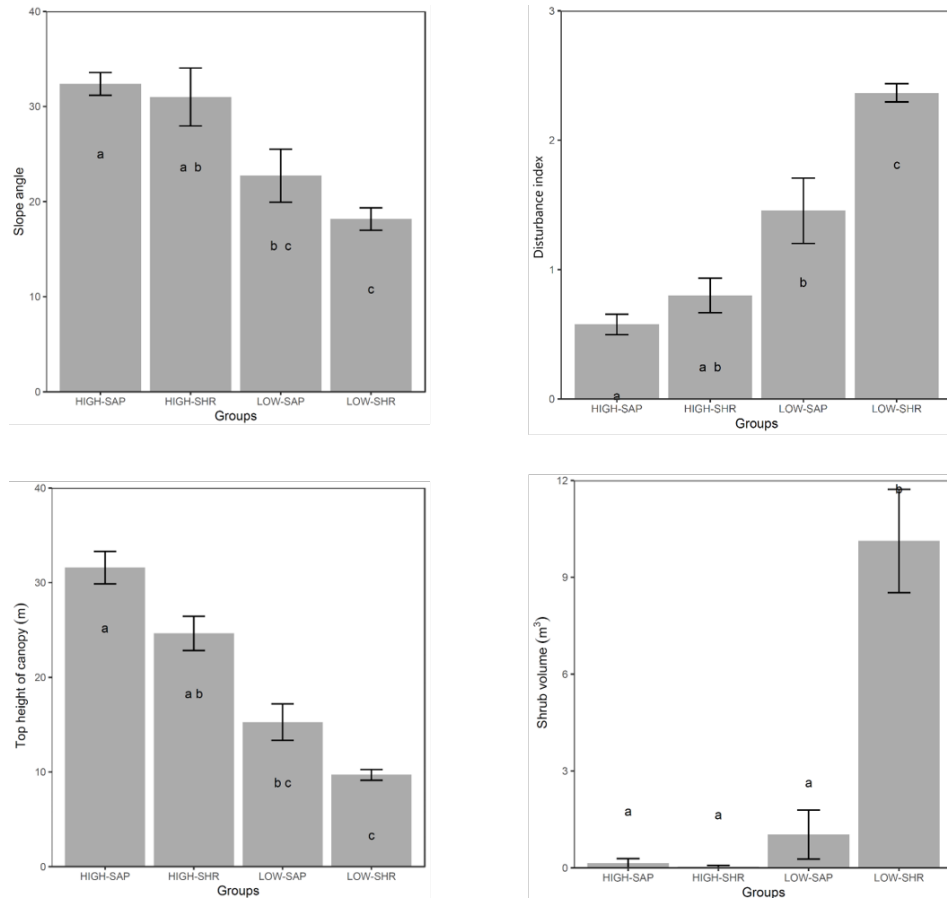


Figure III.4: Selected forest stand and environmental variables averaged for the five habitat clusters. Letters indicates groups that do not significantly differ (Kruskal-Wallis non parametric test, p value < 0.05). Error bars are $\pm$ SE.

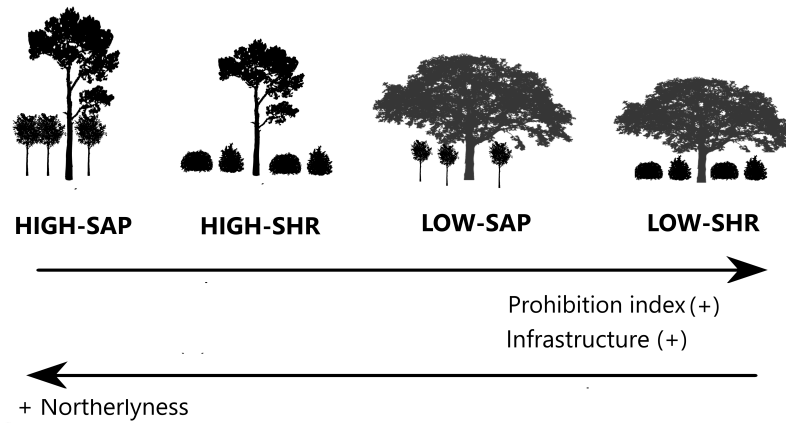


Figure III.5: A figurative representation of the final four woody plant groups. HIGH-SAP - high-canopy forest with an understorey dominated by saplings of tree species; HIGH-SHR - high-canopy forest with an understorey dominated by shrubs; LOW-SAP - low-canopy forest with an understorey dominated by saplings of tree species; LOW-SHR - low-canopy forest with an understorey dominated by shrubs. The main significant explanatory variables of the ordered logit model are shown by the arrows. The direction of the arrow explains if an increase of the variable of interest implies an increase (or decrease) of the probability that the patch is high canopy-sapling forest, rather than low-canopy shrub forest. An increase in the prohibition index points to forest prohibition regimes allowing higher levels of disturbance (increased wood removal and grazing).

Table III.2: Ordered Logit model showing the relationship between selected environmental variables and the final reconfiguration of four woody plant groups across six sacred forests (HIGH-SAP, HIGH-SHR, LOW-SAP, LOW-SHR). The numbers in parenthesis represent the different levels of the ordinal categorical variables.

Explanatory variables	Model coefficient	z value	Odds ratio	95% CI	p value
Northerly aspect	1.21	3.25	3.34	1.61; 6.93	0.001
Soil depth (=1)	-1.13	-1.47	0.32	0.07; 1.46	0.143
Soil depth (=2)	-0.20	-0.27	0.82	0.18; 3.62	0.791
Slope angle	0.02	0.90	1.02	0.97; 1.08	0.369
Disturbance index	-2.13	-3.09	0.12	0.03; 0.46	0.002
Infrastructure	-1.14	-2.22	0.32	0.12; 0.88	0.027
Latitude	0.10	2.41	1.10	0.49; 2.49	0.809
Cut = 1 ^a	39.8	0.22	-	-	-
Cut = 2 ^a	41.8	0.23	-	-	-
Cut = 3 ^a	42.6	0.23	-	-	-

^a Cut points representing the three different levels of the ordered logit model.

III.4 Discussion

III.4.1 Vegetation variability in sacred forests

The first objective of this study was to distinguish the groups of woody plants occurring within sacred forests in northern Greece, based on their tree species composition and structural characteristics. This is important because, up to now, the vegetation composition of Greek sacred forests has not been well researched. Previous studies have focused on the overall biodiversity (flora and fauna) value of such forests compared with non-sacred forests (Avtzis et al., 2018), or on their history, ethnography and folklore (Stara et al., 2016).

Our results show that the vegetation of the sampled sacred forests differed greatly amongst the sites. This finding is not itself surprising as the sites cover a wide range of environmental conditions such as altitude, geology and aspect, which are known to influence species composition. We examined both the canopy and understorey composition separately and together to further understanding of forest structure and dynamics. The species rank analysis revealed that the canopies were dominated by distinct tree species, which were also similar in abundance in the understorey of all six sites. We took this as an indication that the forests are not undergoing a trajectory of change in the composition of their dominant species.

Nevertheless, there were some notable site-specific exceptions, with species in the canopy underrepresented or missing from the understorey. Lack of regeneration of the dominant canopy species has been observed as a common feature of sacred forests that are heavily disturbed by human activities, such as firewood collection, timber extraction and grazing (Mishra et al., 2004; Upadhaya et al., 2008). However, historical evidence shows that Greek sacred forests have been subject to cultural abandonment since ca. 1950, linked to prevailing patterns of rural depopulation (Stara et al., 2015b; Blondel et al., 2010). Where minor discrepancies occurred between canopy and understorey species composition (e.g. in Vitsa, Mazi, and Greveniti sacred forests, Fig. III.1), we link

this to both anthropogenic and environmental factors, possibly indicating (i) former anthropogenic disturbance events, which affected seed dispersal, plant establishment and growth at a given moment in time, (ii) expansion of the forests into the surrounding matrix of former farmland, after it became subject to less active management. Field evidence and historical reconstructions corroborate these processes. In Mazi the discrepancy between species abundance in canopy and understorey layers can be linked to past disturbance events, specifically the lack of medium-sized oaks has been linked to former grazing practices, which reduced the abundance of the most palatable species, e.g. via browsing and cutting small trees to provide fodder (Tsiakiris et al., 2019). In Greveniti, the lack of several canopy species from the understorey (e.g. beech saplings coupled with adult oaks and black pines) seems to indicate that the beech forest has expanded into adjacent areas, which had a different vegetation layer. This has been verified by interviews with local informants and in situ observations.

III.4.2 Management regimes influence the structural and species composition of sacred forests

The final configuration of plot and sub-plot clusters into four woody groups indicates a series of variables significantly associated with tree species composition and structure in the sampled sacred forests: northerly aspect, forest prohibition regimes, and presence/absence of human infrastructure. We can thus infer two main types of variation associated with tree species composition and structure of sacred forests in north-western Greece: (i) variation in site environment (especially aspect); (ii) variation in management and prohibition regimes. Previous studies have identified how site aspect can influence water availability in Mediterranean forests, and that north-facing slopes tend to support taller-stature and denser woody plant communities than south-facing slopes (Kutiel, 1992; Sternberg and Shoshany, 2001). This is reflected in the difference we found in tree height across the four forest communities, which is likely to be linked to the observed variation in floristic diversity (as in Zilliox and Gosselin, 2014). However, we also found evidence of independent variation linked to differences

in the past management regimes amongst the sites associated to their sacred status (forest with strict prohibition regimes versus forest where regulated management practices were permitted, e.g. low intensity grazing), as well as variation in the extent to which these regimes have been maintained to the present day. The logit model shows a significantly strong association of the strictness of the historical prohibition regime and its maintenance, with current tree composition. Similar results were provided by (Pion, 2014)'s earlier ordination work on agropastoralist sacred forests, where different vegetation communities (i.e. open forests, dense forests, and shrublands) were found to be possibly linked to current and former grazing practices.

Through our analysis, sites emerge as being usually dominated by one of the four woody vegetation groups HIGH-SAP in Greveniti, Vovoussa and, partially, Palioseli, LOW-SHR in Vitsa, Mazi and Kapesovo. The dominance of these two groups in the six sacred forests corresponds to the former management regime of the sites: complete protection for slope stability allowing full rein to ecological processes in the former, open canopy managed for shade and grazing in the latter.

The sites dominated by the HIGH-SAP group are all on steep slopes in elevated areas and exhibit minimal indicators of human interference (i.e. no evidence of former or current grazing, limited deadwood extraction). Their condition reflects the strong priority placed on these forests to protect downslope valley settlements from landslides and rockfalls (Volkwein et al., 2011). In addition, sacred forests characterized by these woody groups (at Palioseli and Greveniti) were governed through the threat of excommunication of trespassers, which acted as a strong deterrent to tree cutting or grazing (Stara et al., 2016). Such rules lead to a strict protection regime, which explains the lack of indicators of grazing and of human infrastructure (e.g. churches, roads, fences, etc.) in the HIGH-SAP group. Observance of these prohibitions extended to banning removal of fallen trees at least in Greveniti, which continues to the present day as revealed by Cullen (2015)'s observation of a great number of fallen trees at various stages of decay.

The three sites dominated by the low-canopy forest group (LOW-SHR) are characterized by the presence of browsed shrubs, coppice stools and pollarded trees. This reflects use of the oak and mixed broadleaved communities for livestock grazing, for harvesting of tree fodder, and as shaded rest areas for the animals (in Greek "*stálos*"; Stara et al., 2016, although this now occurs less frequently (Pion, 2014). Thus, the high density of shrubs in these groups is a likely indicator of continuing livestock management practices. Grazing pressure can also influence species e.g. *J. oxycedrus* a species avoided by goats and sheep, which can persist for decades in the understory, and is a good indicator of past and continuing grazing pressure (Rackham, 2014); *J. oxycedrus* accounts for 11.7% of the total number of woody plants sampled in the sub-plots in the LOW SHR group, (Table III.1).

The intermediate woody plant groups HIGH-SHR and LOW-SAP show the strongest evidence of an undergoing change as the legacy of past management diminishes. This can be seen through processes of growth of the understory forest layer and regeneration in canopy gaps. Sites strictly protected to prevent erosion and landslides (which tended to be dominated by the HIGH-SAP group) have expanded through natural regeneration into the adjacent matrix of formerly more open areas. These younger secondary forests are often classified into the HIGH-SHR group. Onsite observations of the vegetation at Greveniti indicates that the high forest has expanded into former wood pasture dominated by deciduous oaks, the remnants of which are still present as large decaying trunks. This is also evidenced by the abundance of *J. oxycedrus*, which accounts for 27.6% of the total number of woody plants sampled in the sub-plots in the HIGH-SHR group (Table III.1). It is often senescent indicating that it is remnant of former open shrubland dominated by heavy grazing impacts.

In areas where the intensity of grazing by goats and sheep has diminished, there has been an increase of tree natural regeneration in open patches identified as LOW-SAP, a likely indicator of a drift towards a less disturbed forest structure. As shown by previous work

at Mazi and Vitsa by Pion (2014), the presence of active grazing has a strong influence on the structure of the understorey — active grazing favours shrub species and shrubby form in palatable tree species (*Carpinus orientalis*) and sclerophyllus species (*Juniperus spp.*), while cessation of grazing allows sapling development.

III.4.3 Conclusion

Through an analysis of vegetation patterns, we have shown that: (i) variation in vegetation structure and composition amongst sacred forests in Epirus is substantial and it is linked to both environmental and socio-cultural drivers; (ii) there is no major evidence indicating an overall trajectory of change in tree species composition between canopy and understorey layers, though there are localized shifts in species and structure related to changes in intensity of management; (iii) distinct woody plant groups can be identified across sites according to key environmental gradients and their association with the intensity and type of past forest prohibition regimes. In particular, historical control of grazing is identified as a major determinant of the current species and structure of sacred forests. Small size, large variability in species composition, and high variation in past prohibitions and the extent to which they have been gradually abandoned, of the sacred forests of Epirus have led to a high level of variation in structure and species composition between sites.

These findings provide a first categorization of Greek sacred forest based on their structural composition at both canopy and understorey level. This analysis proved to be a useful application for understanding the effect of anthropogenic influence on the top of environmental variability. Further applications of this study include: (i) enlarging the sample by including additional sacred forests from neighbouring areas, (ii) comparing the results with adjacent forests with similar composition but where prohibition regimes are absent. The results from this study feed into the model interface deployed in Chapter VI allowing to characterize the vegetation structure of the simulated forest under different social changes.

CHAPTER IV

**Combining tree rings and stand dynamics to identify
common historical patterns of sacred forests in
Northern Greece**

IV.1 Introduction¹

Spatial and temporal dynamics are common features of socio-ecological landscapes of the Mediterranean basin. Dramatic societal changes in rural areas over the past two centuries generated substantial land use changes which had a major effect on forest cover. Well-known examples includes the collapse of transhumance practices and decline of small pastoralist societies (Perez and Saez, 1990), alterations in the form, function and meaning of "traditional management" of agricultural landscapes (Kizos and Koulouri, 2006; Terkenli, 2001), and agricultural abandonment as a driver of changes especially through increasing fire occurrence (Koutsias et al., 2012; Moreira et al., 2001). Any decline in socio-demographic drivers will be reflected by increasingly ecological processes, which may begin to dominate land cover, e.g. through vegetation succession on old fields, vineyards and orchards (Zomeni et al., 2008). The conservation significance of these dynamics can be mixed as although forest expansion may encourage greater diversity of forest-dwelling organisms, it may also impoverish taxa requiring more open spaces (Tsiakiris et al., 2019; Bugalho et al., 2011; Alrababah et al., 2007).

Understanding the history of a landscape thus becomes a complex and multidisciplinary exercise, when the environment under study is a long-term cultural landscape where ecological and social factors are strongly interrelated (Farina, 2000). This is especially the case for sacred forests and groves. Local indigenous knowledge and peoples beliefs, economic and demographic drivers, dominance and autecology of tree species are all relevant factors in the trajectories a socio-ecological system can take. The overarching aim of this thesis is to disentangle the relative influence of ecological and social processes in the composition, spatial patterning and temporal extent of Sacred Natural Sites. To this end, apprising how temporal changes in cultural and socio-economic drivers are

¹The analysis and results of this chapter have been presented in one scientific conference (Trace Conference, Bialowieza, Poland, 2016; oral presentation with title *Evidence from tree rings and forest history of sacred groves in Northern Greece*).

reflected in the ecological layer of a pool of sacred forests in Epirus allows us to better understand present tree composition and assess future trends with different management scenarios essential to rational management planning. This is even more important when sacred forests are threatened by cultural abandonment — as in the case of sacred forests in Epirus (Greece) which have, until recently, been given little attention outside their immediate communities.

The guiding hypothesis for this chapter is that the long term structural development of sacred forests is determined by socio-economic drivers acting on ecological processes. In particular, we aim at verifying whether we can associate distinct waves of recruitment of canopy trees to common regional socio-economic changes, mainly human population trends. Our research builds on the available historical knowledge of the forests by linking it to empirical evidence on the current forest structure and composition, which is obtained through a detailed forest inventory of five sacred forests. Specifically, we investigate the tree population dynamics using a combination of tree aging through dendrochronology and assessment of current tree population size-class structures. Subsequently, we link estimated ages for the canopy tree layers to human population data in the areas. Emerging patterns across sites are identified and discussed. This analysis allows us to (i) validate available oral and historical narratives, (ii) further explore the functional role that these forests played for the local communities, and (iii) identify common patterns and features among forests that can be used as an indication of the general functional role of sacred forests at the regional level.

IV.2 Material and methods

Five sacred forests were selected for the present study. These were: (i) *Agios Nikolaos* in Livadakia, belonging to the village of Vitsa (ii) *Kouri*, village of Mazi; (iii) *Mereao*, village of Palioseli; (iv) *Agia Paraskevi*, village of Vovoussa; and (v) *Toufa*, village of Greveniti. Kapesovo sacred forest was excluded from the sample due to the bad quality of the tree cores there extracted. More information on the forests is presented in Chapter II.

IV.2.1 Tree size-class distribution

Tree size-class distribution within a forest is commonly used as an indicator of forest structure, and to infer the effect of past disturbance events (Lai et al., 2013; Burkhart et al., 2012). To develop such curves for the study forests, data arising from the forest inventory described in Chapter II was utilised. In particular, the diameter at breast height (1.3 m) for the main dominant tree taxa in each of the forest was computed. The dominant tree taxa in each forests were identified as *Quercus spp.* in Vitsa and Mazi, *Pinus nigra* in Vouvousa and Palioseli, *Fagus sylvatica* in Greveniti (as presented in Chapter III). Size class distribution plots were obtained for the dominant taxon in each forest and an aggregation of the remaining taxa. All size class distribution were finally fitted to several theoretical fitting choosing among the following probabilistic distributions: linear, log-normal, log-logistic and polynomial. Both the coefficient of determination R^2 and the Akaike Information Criterion (AIC) were used to identify the best fitted model amongst them.

Additionally, trees of the dominant tree taxa listed above in each forest were cored at a height between 0.6 and 1 m, as to age specific tree cohorts and explore past growth dynamics. In the pine woods, the tree with the largest diameter in each plot was selected for coring, whereas in the oak and beech forests the second largest individual was selected, as a high proportion of the largest trees were found to have cavities in their centre . Also, we avoided sampling the largest trees as this data was already collected and available within the project THALIS-SAGE dataset for 15 sacred forests (Kyparissis et al., 2016). In a second stage of sampling, additional individuals were selected for coring to fill in gaps in the diameter size range of cored trees (with diameter at breast height $\geq 5cm$). This resulted in a total of 221 individual cores for five study areas available for further analyses². Additional information on the sampling protocol is

²204 cores extracted personally with the field work assistance of N. Pion, and R. Cullen, and 17 available from the THALIS-SAGE project).

provided in Chapter II.

IV.2.2 Estimation of tree ages from cores

Tree age was estimated by counting rings in the cores backwards from the first ring behind the bark to the pith. In this process, 16 additional cores were discharged, as rotten, damaged, or with rings that could not be distinguished. This reduced the available dataset to 205 cores. As oldest deciduous oaks were often rotten in the middle and often the angle of the core did not intersect the oldest rings in the tree (a major problem which can arise when extracting increment cores; Duncan, 1989), not all the cores included the chronological centre of the tree (i.e. the pith). To avoid repeat coring of trees in such culturally sensitive sites, a methodology to estimate partial and incomplete cores was used. Models to estimate missing rings are widely used in the literature when using partial wood cores (e.g. Frelich and Graumlich, 1994; Duncan, 1989; Norton et al., 1987). In this study, we followed Rozas (2003)'s assessment to estimate partial (showing the arch of the inner rings) and incomplete (missing the inner arch and the pith) cores (Fig. IV.1 and Table IV.1).

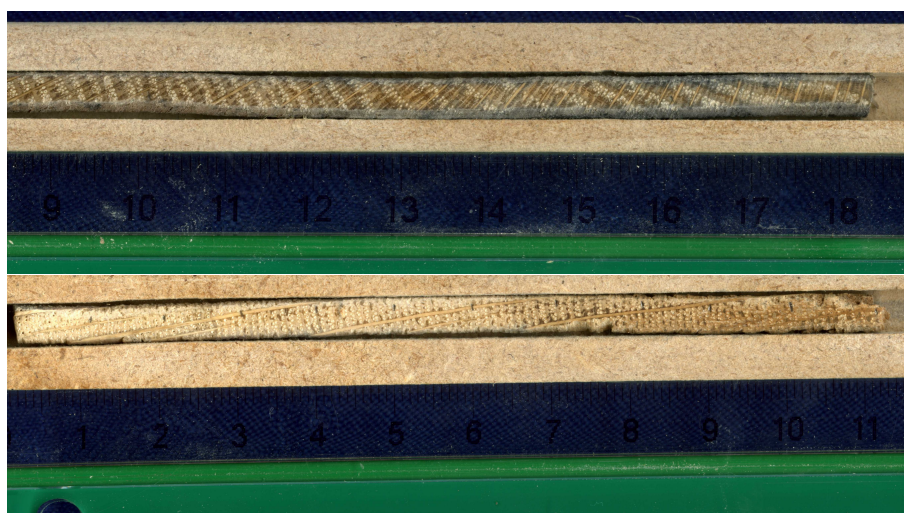


Figure IV.1: An example of partial (top image) and incomplete (bottom) deciduous oak cores. The topmost core does not include the tree pith (i.e. the very centre of the tree), but it shows the curvature of the inner tree ring arch. The bottom core misses as well the inner arch (i.e. the abrupt inner curvature of the first tree rings).

Table IV.1: Number of full, partial and incomplete tree cores sampled in five sacred forests.

Site	Cored species	Full cores ^a	Partial cores ^b	Incomplete cores ^c	Total cores
Vitsa	<i>Quercus cerris</i>	2	19	5	68
	<i>Quercus frainetto</i>	3	10	16	
	<i>Quercus. pubescens</i>	1	3	2	
	<i>Quercus spp. others</i> ^d	1	3	3	
	Total	7	35	26	
Mazi	<i>Q. cerris</i>	2		1	46
	<i>Q. frainetto</i>	3	3	5	
	<i>Q. pubescens</i>	2	6	7	
	<i>Quercus trojana</i>	2	8	3	
	<i>Quercus petraea</i>	1			
	<i>Quercus spp. others</i> ^d			3	
	Total	10	17	19	
Palioseli	<i>Pinus nigra</i>	9	13	3	31
	<i>Q. cerris</i>	5			
	<i>Q. petraea</i>	1			
	Total	15	13	3	
Vovoussa	<i>P. nigra</i>	15	6	2	29
	<i>Q. cerris</i>	4			
	<i>Quercus spp. others</i> ^d	2			
	Total	21	6	2	
Greveniti	<i>Fagus sylvatica</i>	8	18	4	31
	<i>P. nigra</i>	1			
	Total	9	18	4	
		62	89	54	205

^a Cores including the inner pith (i.e. the very centre of the tree).

^b Cores showing the arch of the inner rings (i.e i.e. the abrupt inner curvature of the first tree rings) but not the pith.

^c Cores showing neither the pith nor the arch of the inner rings.

^d Including hybridizations across *Quercus* species, and unidentified *Quercus* spp.

This involved testing eight different methodologies to estimate missing rings. All estimation methods follow Rozas (2003) who based his selection on the most frequent criteria employed in the literature (e.g. Frelich and Graumlich, 1994; Norton et al., 1987). These are:

- Four methods based on average growth of prior (i.e. the last visible) rings, which assume that the rate of radial growth for a given tree is constant over the tree growing period (Equation 1). These methods estimate the age of the missing portion of the core by dividing the length of the missing radius by the Mean Ring Growth (MRG) —the average— of the prior 5, 10, 15 and 20 visible rings (MRG5, MRG10, MRG15, MRG20);
- Four methods based on average basal area increment, assuming a constant Basal Area Increment (BAI; the increment in tree basal area from one year to the next) over the tree growing period (Equation 2). The age estimation of the missing core portion is assumed to be equal to the BAI of the length of the missing radius divided by the mean BAI of the prior 5, 10, 15 and 20 visible rings (BAI5, BAI10, BAI15, BAI20).

The methods for estimating partial and incomplete cores were tested using all available full cores separately for each of the three taxa (*Quercus spp.*, *P. nigra*, *F. sylvatica*). To enlarge the available number of full cores for the age estimation process, additional trees were sampled from forests nearby the five study sites that were as closely matched to them in terms of forest composition and structure.³ This gave a total number of full cores of 32, 31 and 8 for each taxon respectively. For each ring r of the full cores, the incremental length of the core c (L_{cr}), the mean growth rate, and the mean basal area increment of the 5, 10, 15 and 20 prior rings were computed.

³Specifically six pine cores from Konitsa (2), Vrysochori (2), and Molista sacred forests (2); three deciduous oak cores from Kapesovo sacred forest. The location of the forests is provided in Fig.II.1, Chapter II.

MRG methods estimate the missing age of a partial/incomplete core as:

$$mage_c = \frac{G_c - L_{cr}}{MRG_{cr}} \quad (1)$$

where the missing age of the core c is equal to the ratio between the missing radius and the mean ring width of the prior 5, 10, 15, and 20 rings at the given ring r . The missing radius is obtained subtracting from the geometric radius at core height (G_c) the incremental length of the core c at the given ring r (L_{cr}). To obtain the core total age, the above value needs to be added to the age obtained from the respective partial/incomplete core.

Basal area increment was estimated as:

$$mage_c = \frac{BA(G_c - L_{cr})}{BAI_{cr}} \quad (2)$$

where the missing age of the core c is equal to the ratio between the basal area of the missing radius, and the mean BAI of the prior 5, 10, 15, and 20 rings at the given ring r . Similarly to above, the core total age is obtained adding $mage$ to the residual age obtained from the respective partial/incomplete core.

Following the procedure in Rozas (2003) a subset of 500 random rings excluding those in the outer 20% of the core for oak, beech and pine were selected from all available full cores (71 in total). Each of the 500 rings per taxon was assumed to be the innermost available ring of its core, and the number of missing rings was calculated with each of the eight estimation methods. If, for a given core, the estimated length of the missing radius was a negative number (when the core length is greater than the geometric radius of the tree due to eccentric growth), this core was excluded from the analysis. In total 13 partial and incomplete cores were thus removed. Each estimated value was then compared with the real number of missing rings at the selected ring and the difference between these values in absolute terms was used to test which of the eight estimation methods held the significant smallest error via a Kruskal-Wallis test. This analysis was then iterated 50 times.

Mean estimation errors obtained with the BAI methods were smaller in absolute terms than those obtained with the MRG methods (Table IV.2). In particular, median BAI errors were significantly smaller than the MRG ones, both for *Quercus spp.* (Kruskal-Wallis, $\chi^2=199.87$; *d.f.* = 7; $p < 0.001$) and *P. nigra* (Kruskal-Wallis, $\chi^2=813.18$; *d.f.* = 7; $p < 0.001$). However, notable differences were not found between the BAI methods nor amongst the MRG methods. Only for *F. sylvatica* the error from the BAI20 method was significantly larger than for all the other seven methods (Kruskal-Wallis, $\chi^2=26.91$; *d.f.* = 7; $p=0.001$). The BAI10 estimation method was chosen for all species as it produced the smallest error overall (35.8 ± 0.3 years for *Quercus spp.*; 32.8 ± 0.3 years for *P. nigra*, and 28.1 ± 0.1 years for *F. sylvatica*). This is in line with the findings of Rozas (2003) (errors of around 43 years for *F. sylvatica* and 36 years for *Quercus robur*), and by previous studies that estimated age of partial and missing cores (Duncan, 1989; Norton et al., 1987). Using this method, the ages of 89 partial and 54 incomplete cores were quantified. For the whole dataset of 195 analysed cores, the oldest *Quercus spp.* core was 343 ± 36 years old and was found in Vitsa, the oldest *P. nigra* core was 250 ± 33 years old and was in Palioseli, and the oldest *F. sylvatica* core had an age of 198 ± 28 years and was in Greveniti (Appendix I).

Finally, all the aged cores, grouped by site, were used to test for relationship between tree age and diameter at breast height of the dominant taxon using Ordinary Least Square regressions. The linear model assumptions were tested and influential points showing a Cooks distance larger than 0.5 were carefully investigated. If such outliers corresponded to incomplete cores, then they were interpreted as error in the age estimation process and hence removed.

Size class distribution plots of the three dominant taxa (from diameter at breast height ≥ 5 cm) were quantified in yearly ages through predicted values of the age-diameter linear regressions for each site described above. This step allowed to assign an age in years to the different cohorts identified in the forests. These distributions were smoothed using Kernel Density Estimations (KDE). KDE is a non-parametric estimation that allows to

Table IV.2: Absolute errors in tree ages (in years) estimated from a sample of 500 random rings per species (100 for *Fagus sylvatica*) for eight different methods of age estimation. For each method, the mean error is reported together with the results of a Kruskal-Wallis non-parametric test on the median. Methods that do not differ significantly are labeled with the same letter code. The method giving the lowest mean error of tree ages for each species is shown in bold.

		MRG5	MRG10	MRG15	MRG20	BAI5	BAI10	BAI15	BAI20
<i>Quercus</i> spp.	Mean error	65.2	63.5	62.8	62.3	35.9	35.8	36.1	36.5
	($\pm$ SE)	(0.6)	(0.6)	(0.6)	(0.6)	(0.3)	(0.3)	(0.3)	(0.3)
	KW results	a	a	a	a	b	b	b	b
<i>Pinus nigra</i>	Mean error	95.1	94.5	95.9	98.1	34.5	32.8	32.5	32.7
	($\pm$ SE)	(0.7)	(0.7)	(0.7)	(0.7)	(0.3)	(0.3)	(0.3)	(0.3)
	KW results	a	a	a	a	b	b	b	b
<i>Fagus</i> <i>sylvatica</i>	Mean error	34.5	30.1	29.0	28.3	28.6	28.1	28.6	84.7
	($\pm$ SE)	(0.2)	(0.1)	(0.1)	(0.1)	(0.1)	(0.1)	(0.1)	(3.9)
	KW results	a	a	a	a	a	a	a	b

fit a smooth curve to a given set of observations (Sheather, 2004). In particular, the KDE algorithm is computed by weighting the data points distances to a fitting curve, whose smoothness is determined by a free parameters, so called *bandwidth*. By merging trees KDEs and historical village population data (Appendix J), we were able to assess the primary hypothesis of this chapter: whether different waves of recruitment of canopy trees can be traced back to common regional socio-economic changes. All statistical analyses were performed using the statistical language R (R Core Team, 2017).

IV.3 Results

IV.3.1 Population size structure

Figure IV.2 shows the population size class distributions assessed for all species combined and for each of dominant taxon at canopy level for each of the five sacred forests. Figure IV.4 presents similar graphs, but with a focus only to the distribution of the dominant taxa for each of the studied forest. Stand curves for all the five sampled

sites approximate exponential decay curves, with the log-logistic distribution resulting as the best theoretical fit for all the sites producing a mean R^2 of 0.66 ± 0.05 SE across the sites (Appendix K). In the oak sacred forests of Vitsa and Mazi, deciduous *Quercus spp.* (*Q. cerris*, *Q. pubescens* and *Q. frainetto*), with the addition of the semi-deciduous *Q. trojana* in Mazi are present in all diameter classes. Deciduous oaks are the most abundant tree species occurring for size greater than 25 cm, and the only ones occurring in diameter classes larger than 50 cm (45 cm in Mazi). *Quercus spp.* with diameters up to 97 cm were measured. Other tree species not dominant at canopy level (e.g. *Quercus coccifera*, *Juniperus oxycedrus*, *Carpinus orientalis*, *Phillyrea latifolia*) are heavily concentrated in smaller diameter classes, not exceeding 25 cm in size. In pine sacred forests (Vovoussa and Palioseli), *P. nigra* individuals dominate the top canopy layer. Pines are the most abundant tree species occurring for size greater than 30 cm in both sites, and the only ones occurring in diameter classes larger than 85 cm (deciduous oaks of large diameter classes are also sparsely present in the forests). The largest black pine was measured in Vovoussa, with a diameter at breast height of 118 cm. Like in oak sites, smaller diameter classes are mostly dominated by other tree species (*C. orientalis*, *Carpinus betulus*, *Ostrya carpinifolia*). Lastly, in the beech forest of Greveniti, *F. sylvatica* is the dominant species both at canopy and understorey level. Beech trees are the most abundant tree species occurring for size greater than 45 cm, and the only ones occurring in diameter classes larger than 100 cm. The largest sampled beech measured 137 cm in diameter.

As expected, there is a significant positive relationship between diameter at breast height and age for all sites, with model coefficients of determination (R^2) all above 0.4 (Fig. IV.3). Results highlight a large variation in growth rate amongst the canopy trees in some sites, namely Greveniti, Mazi and Palioseli reflected in lower R^2 (respectively 0.4, 0.5, and 0.55). Additionally, the graphs show a large variation in tree growth rate between the sites (i.e. the gradient of the regression lines). The average growth rate was fastest for the beech forest (Greveniti) and slowest for the oak sites (Mazi and Vitsa) with Pine

forests (Vovoussa and Palioseli) showing intermediate values.

Figure IV.2: Tree diameter size-class distributions in the five study forests distinguishing between canopy species, other tree species and dead trees. The main canopy species are: Vitsa (*Quercus* spp. *Q. cerris*, *Q. frainetto*, *Q. pubescens*); Mazi; (*Quercus* spp.: *Q. cerris*, *Q. frainetto*, *Q. pubescens*, *Q. trojana*; Greveniti (*Fagus sylvatica*); Vovoussa (*Pinus nigra*); Palioseli (*P. nigra*). Size classes are defined in 5 cm classes by the lower interval limit. The list of the remaining tree species is provided in Appendix H.

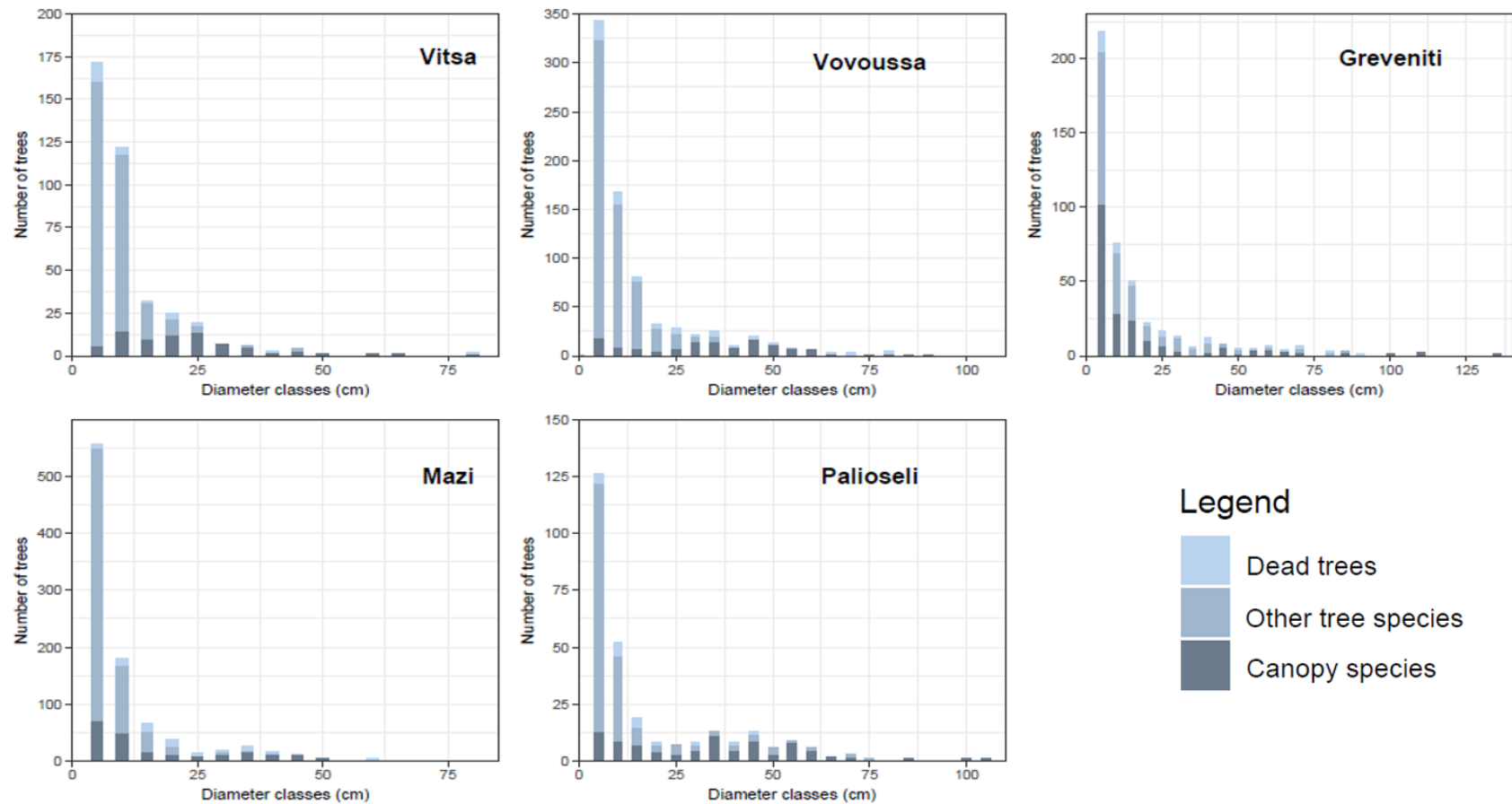
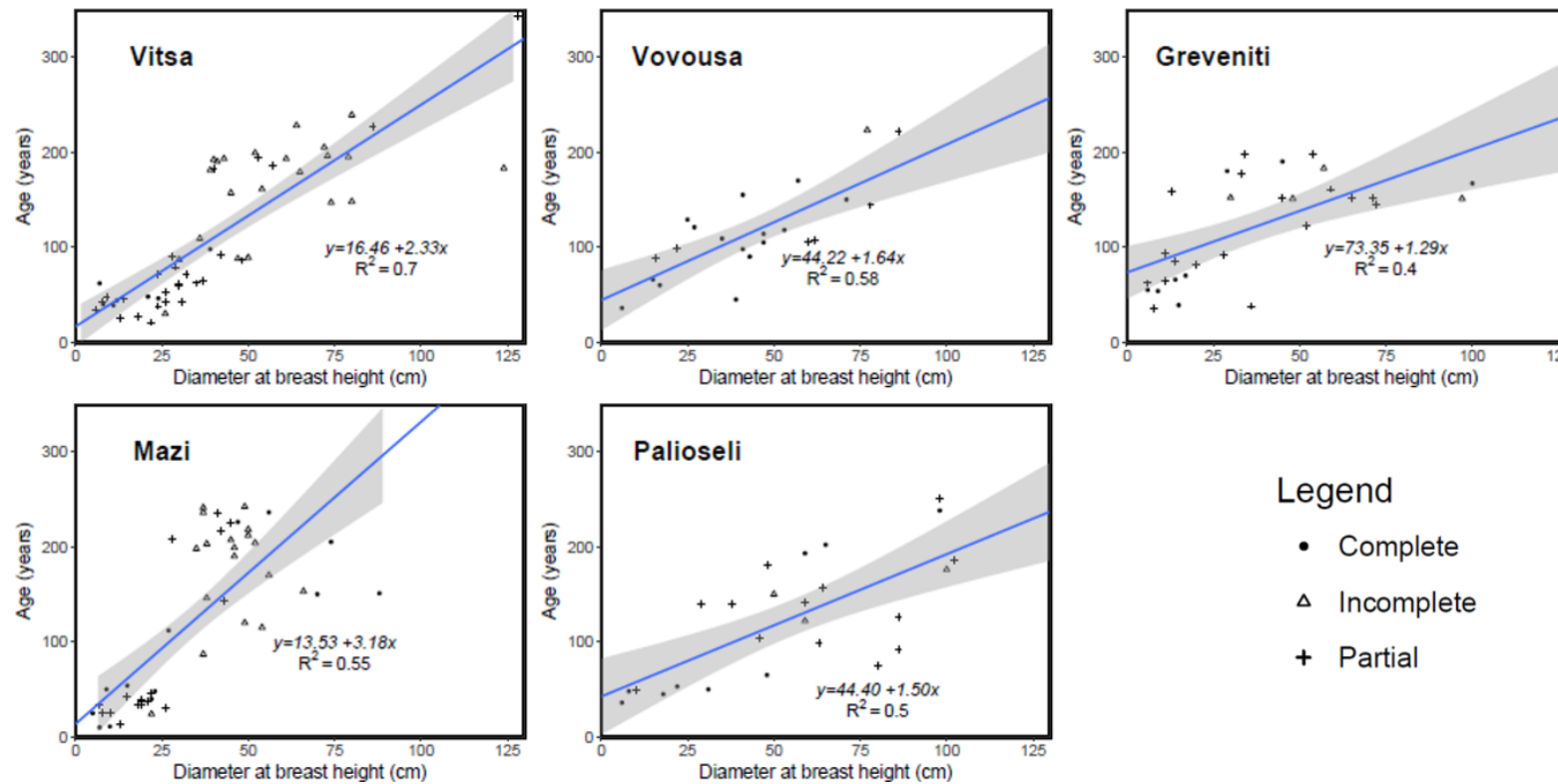


Figure IV.3: Age-diameter at breast height (1.3 m) relationships and regression lines for cored trees in the five study areas: Vitsa, Mazi; (*Quercus* spp: *Q. cerris*, *Q. frainetto*, *Q. pubescens*, *Q. trojana*); Greveniti (*Fagus sylvatica*); Vovoussa; Palioseli (*Pinus nigra*). 95% confidence bands around each regression line are shown by the grey shading. The cores that were complete or for which ages were estimated from incomplete or partial cores (defined in Table IV.1) are shown by separate symbols. All five regressions shown a significant and positive relationship between diameter at breast height and age (Vitsa : 2.33; $t = 11.15$; $p < 0.001$; Vovoussa 1.64; $t = 2.92$; $p < 0.001$; Greveniti 1.29; $t = 4.33$; $p < 0.001$; Mazi 3.18; $t = 7.44$; $p < 0.001$; Palioseli 1.50; $t = 4.84$; $p < 0.001$).



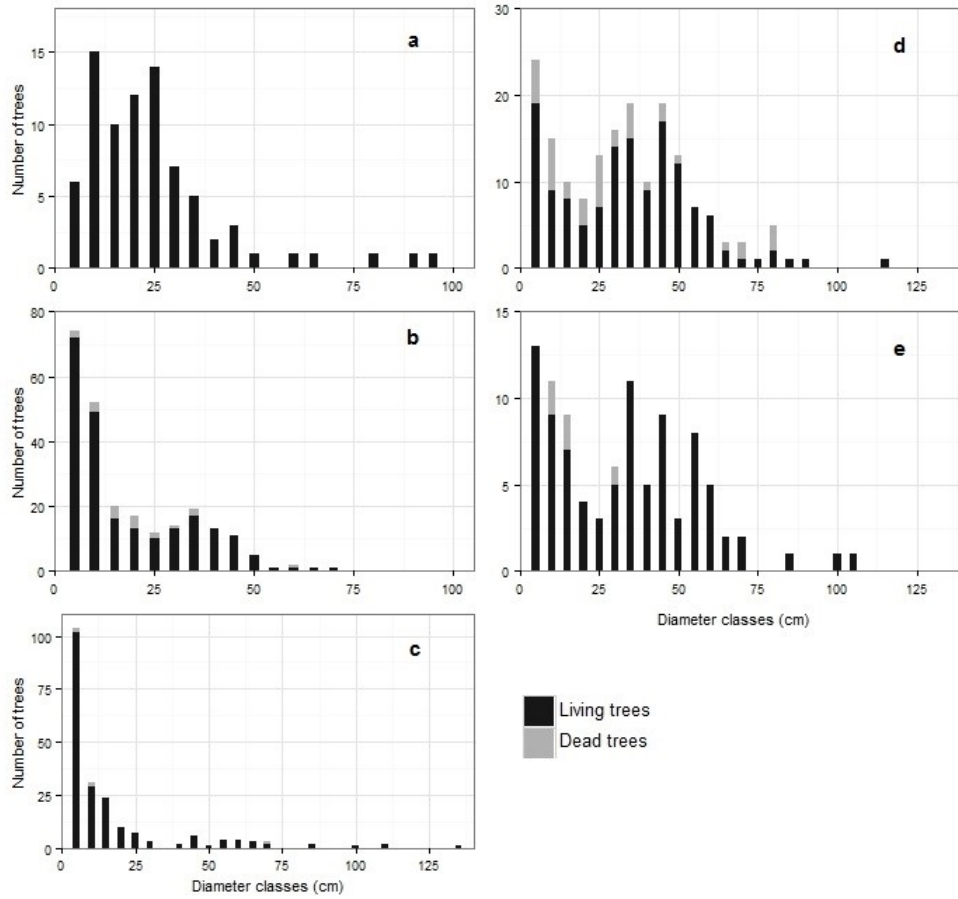


Figure IV.4: Tree diameter size-class distributions of the main canopy species in the five study forests. (a) Vitsa (*Quercus* spp. *Q. cerris*, *Q. frainetto*, *Q. pubescens*); (b) Mazi; (*Quercus* spp.: *Q. cerris*, *Q. frainetto*, *Q. pubescens*, *Q. trojana*); (c) Greveniti (*Fagus sylvatica*); (d) Vovoussa (*Pinus nigra*); (e) Palioseli (*P. nigra*). Size classes are defined in 5 cm classes by the lower interval limit.

IV.3.2 Timing of tree recruitments

Figure IV.5 shows the kernel densities obtained from the ages of all surveyed trees belonging to the three dominant taxa for the five sites⁴. The relationship between age and diameter was estimated from the linear regressions presented in Fig. IV.2. Results show comparable tree establishment events across sites:

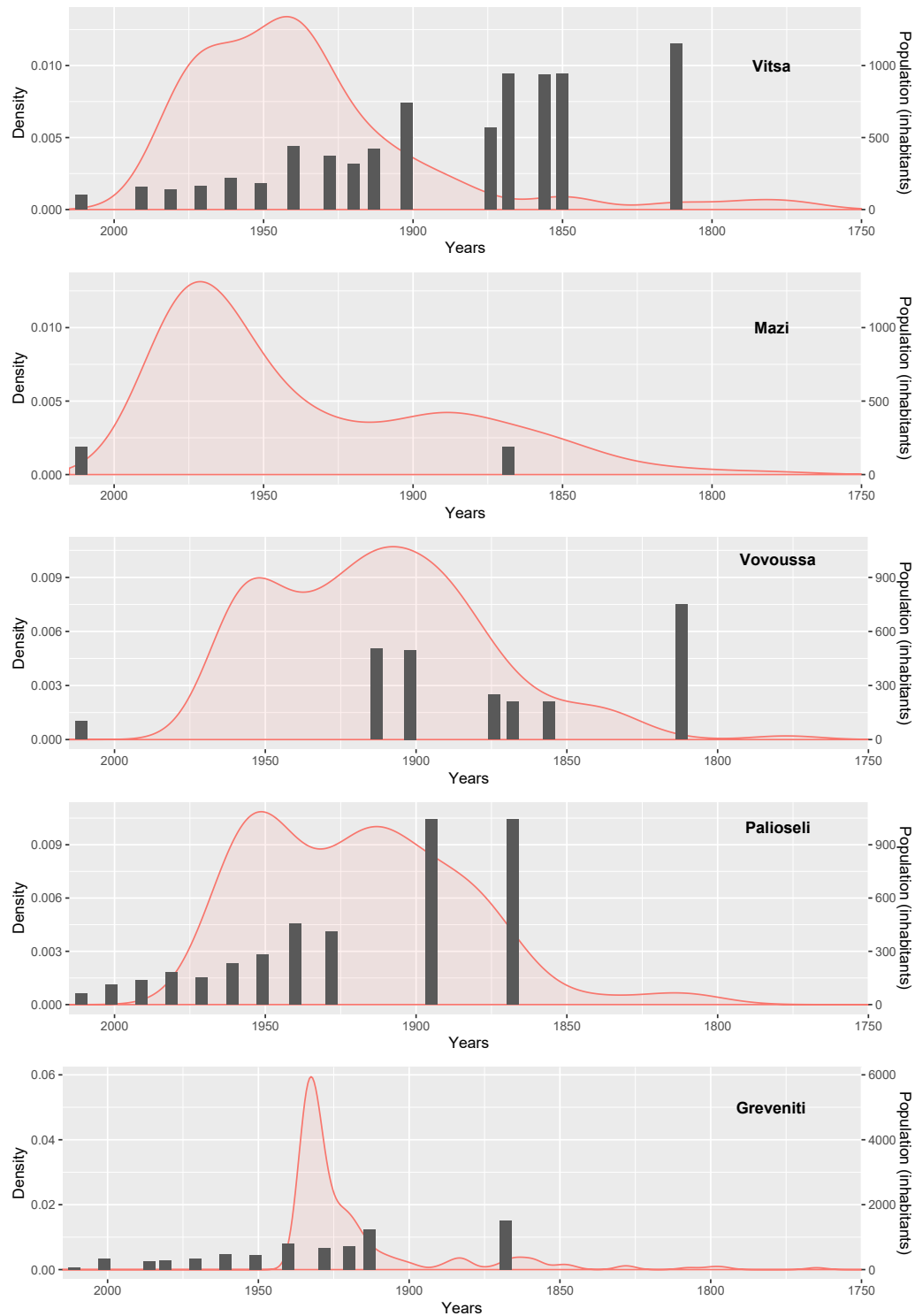
⁴The bandwidth parameter was estimated using the Silverman's rule-of-thumb imbedded in R. This rule chooses the parameter as 0.9 times the minimum of the standard deviation and the interquartile range divided by 1.34 times the sample size to the negative one-fifth power (Silverman, 2018). The corresponding results for the five KDEs are: Vitsa 4.82; Mazi 4.45; Vovoussa 6.86; Palioseli 8.23; Greveniti 2.54.

- Oak dominated sites are characterized by two density peaks: one of trees established in the second part of the 20th century (between 1965 and 1975 in Mazi, and between 1940 and 1955 in Vitsa), and an earlier one occurring in the first half of the 19th century (between 1840 and 1890 in Mazi, and 1765-1865 in Vitsa), mostly visible in Mazi.
- Pine dominated sites are characterized by three density peaks: one of trees established around 1940, another one in 1890-1915, and a last one in between 1815 and 1865 (1790-1840 in Palioseli), mostly visible in Vovoussa.
- The beech dominated site (Greveniti) is characterized by a unique peak of trees established around 1940. Peaks of smaller intensity are present dating back to 1865 and 1890 years respectively.

By comparing trends in tree density with human population at village level⁵ (Fig. IV.3, secondary Y axis), a correlation between periods of tree recruitment and drops in human population levels is apparent. These relationships are mostly visible for Vitsa, Vovoussa and Palioseli. In particular the 1940-1955 peak in Vitsa is foreshadowed by a population drop of 43.1% (between 1912 and 1913); the 1815-1865 peak in Vovoussa by a population drop of 72% (between 1812 and 1856); the 1940 and 1890-1915 density peaks in Palioseli are associated with a population drop of respectively 60.4% (between 1895 and 1928) and 38.9% (between 1940 and 1951).

⁵Census population flows were adjusted to exclude non-residents inhabitants, using historical and ethnographical resources (Appendix J).

Figure IV.5: Kernel densities (red curves) of the estimated ages for the three dominant taxa in the five study areas: Vitsa, Mazi; (*Quercus* spp: *Q. cerris*, *Q. frainetto*, *Q. pubescens*, *Q. trojana*); Greveniti (*Fagus sylvatica*); Vovoussa; Palioseli (*Pinus nigra*). Tree age was estimated only for diameters at breast height ≥ 5 cm. On the secondary y-axis, population data at village level (Black bars; sources available in Appendix J). X-axis indicates years of tree establishment and relative human population data.



IV.4 Discussion

IV.4.1 Determinants of change in forest population structure

All study sites show an overall decline in tree numbers with increasing diameter, with the stands exhibiting a negative distribution curve, typical of moderately undisturbed forests (Pulido and Díaz, 2005; Condit et al., 1998). However, this ordered reduction in tree density with size is not apparent when assessing the individual tree species size class distributions. *Quercus spp.* in Vitsa and Mazi, and black pines in Palioseli and Vovoussa have a bimodal distribution, peaking around their modes (Fig.IV.4). This suggests that the main canopy species in four of the five sites, show evidence of periods of lower tree recruitment rates, occurring between 50 (Mazi), 80 (Vovoussa and Palioseli) and 100 years ago (Vitsa) (years 1915-1965). This might indicate an historical recruitment failure possibly resulting from increased human pressure which is similar to what has been found for dehesa wood pastures in Spain (Pulido and Díaz, 2005). In the study areas, the lack of recruitment is probably linked to active grazing and trampling in pastoralist sites (Tsiakiris et al., 2019) and increased cutting of small trees by trespassers in pine sites. As canopy species decline in numbers, shrubby and understorey species (e.g. *Q. coccifera*, *J. oxycedrus*, *C. orientalis*, *P. latifolia*) increase in abundance, even if confined to the smallest diameter classes (Fig. IV.2). Historical evidence shows that the understorey species were formerly kept low by local management practices, such as active grazing in Vitsa and Mazi (Pion, 2014). With the progressive reduction in human activities in the area and change in management practices, they expanded in size increasing the competition with the main canopy trees samplings (Pulido and Díaz, 2005).

An exception of this behaviour is the beech forest of Greveniti, where the distribution of the canopy species follows an exponential decay curve. This difference with the oak and pine forests can be explained by the fact that in Greveniti the excommunication forest ritual protection developed to conserve the forest because of its valuable physical

protective function (Chapter II), might have provided stronger protection from human trespassing and thus contributed to generate a quasi-natural environment.

We estimate that the largest stand of canopy trees in the five forests are around 225-250 years old. This generally fits with available historical reconstructions. Sacred forests are believed to have been established during the Ottoman occupation (1479-1912), being related with settlements that are now ruined or with others that coagulated into the location of present villages during the 16th and 17th centuries (Lambridis, 1870). The remaining large trees could then be part of the first generation of trees constituting the sacred forests.

In addition, pines and oak sacred forests present a number of large trees with a former open-grown structure (as found in Mazi; Fig. IV.6). This suggests that the trees could have occurred at a lower density during the early stages of sacred forest establishment, similarly to wood pasture elsewhere in Europe (Bergmeier et al., 2010). This matches the historical and ethnographic evidence collected on the sites, indicating that both pine and oak forests were formerly constituted by a greater amount of open areas (Tsiakiris et al., 2019). Moreover, Stara et al. (2016) indicate that many forests were sacralised to protect wood reserves and wood pastures, using the big trees as livestock shelters from the summer heat. These open spaces have now progressed into denser forests as a result of natural regeneration linked to the abandonment of grazing in oak forests (Vitsa and Mazi), or to the reduced use from the local communities of the pine forests (Palioseli and Vovoussa). This former open landscape is also indicated by site-specific features such as the presence of decaying *Juniperous spp.* individuals (light-demanding species), suggesting that they originated in a much more open landscape (Marks and Gardescu, 2001). The high density of large size individuals identified onsite also indicates that traditional management practices aiming to preserve the canopy layer, as reported by historians (Stara et al., 2015a), proved to be successful in preserving the initial cohort of trees in the forests.

Figure IV.6: Two examples of formerly open grown deciduous oaks surrounded by regrown understorey in Mazi sacred forest. ©Marini Govigli V.



IV.4.2 Recurring recruitment periods in the forests

By aging specific emergent trees, we can observe two common recruitment periods across the five studied sacred forests. The first one identified is between 1840 and 1915. This period coincides with an era of human population decline mainly driven by the combination of two historical events: (i) the effect of a plague (the 1812-1819 and 1835-1838 Ottoman plague epidemics, the last strand of the Black Death pandemic hitting Europe from the 14th century; Agoston and Masters, 2009), and (ii) the turbulence and insecurity due to the last period of the Ottoman Occupation. Both phenomena contributed to a large population decline as people moved to Athens, but also to the USA and Africa, reducing the population in the area by 30% from 1873 to 1902 (Damianakos et al., 1997; Papageorgiou, 1995). Additionally, during the first years of the 20th century, historical evidence shows how sacred forests were also gradually abandoned of as a result of the weakening role of the communities and the Church due to the transition of authority to the new Greek State (1913).

Another shared tree establishment period was identified from the tree age data to the

mid-20th century, from 1940 (Vitsa, Palioseli, Vovoussa and Greveniti) to 1975 (Mazi). This period includes two major historical events happening in the region: the Second World War (1940-1944), and the subsequent Greek Civil War (1946-1950). These tree establishment events could correspond to the gradual but progressive afforestation arising from the abandonment of rural areas, due to wars and the subsequent rural exodus that characterized the whole rural economy in Greece (Damianakos et al., 1997). The collapse of local economies had a major impact on the wood pasture sacred forests (i.e. Vitsa, and Mazi), directly related to changes in grazing practices (Papanastasis, 2007). The human population decline and the subsequent afforestation of rural areas might have had an impact on the extensions of the sacred forests. Further analysis of the forest spatial structure will help to investigate this topic further, by showing whether the historical, ecological, and social boundaries of the forests have changed over time.

IV.5 Conclusion

The results from this chapter helped to shed light on the temporal establishment of sacred forests in northern Greece, mapping distinct waves of recruitment of canopy trees within the forest history, and linking them to common regional socio-economic changes and shared historical events. Understanding common patterns become even more critical in the case of "endangered" cultural landscapes in which local knowledge and traditional management practices are fading, due to historical and demographical changes.

In particular, we have provided evidence that, four of the studied sacred forests (except Greveniti) were formerly made of a greater amount of open areas which have now progressed into denser forests because of natural regeneration due to a reduction of human population which reduced pressures of grazing and trespassing. Remaining large trees, dating from around mid-18th century, are likely to form part of the first generation of trees constituting the sacred forests. Also, this study provides evidence of how the changes in the relationship with local communities have had a major effect on the forest composition of such sites. All sampled sacred forests went through similar

tree-cohort establishment phases linked to depopulation flows and events of anthropogenic turbulence and insecurity, as documented in local history.

These findings highlight the strong coupled interactions between human and ecological components of Northern Greece sacred forests. Under the current conditions of reduced human impact, due to major socio-economic changes (mainly land abandonment), the changing relationship of local communities with their surrounding landscape has transformed ecologically some of the sacred forests. Such findings are valuable for better understanding the responsiveness of the forests to drastic exogenous societal changes, as those occurred in the study region during the last 200 years, and they will be further explored in the next chapters by: (i) identifying how trees in sacred forests distribute spatially, as a way to better interpret the forests' boundaries structure, (ii) developing a model of forest expansion based on a Lefkovitch size-structured model of population growth, through which we can estimate quantitatively how sacred forests react to changing socio-economic processes. The latter is particularly important especially when the regulative environment is moving towards the conservation of such sites, as their intrinsic dynamism can challenge the simple concept of preservation as a basis for forest protection.

CHAPTER V

**Where to draw the line: the role of cultural
perceptions in boundary formation of Epirus sacred
forests**

V.1 Introduction¹

A place can be defined as “*a particular position, point, location, a portion of space*” that can be “*designated or available for or being used by someone*” (OED, 2020). This definition implies that places are characterized by a set of traits, which make them definable in space and available for use by people and communities. A place is made up of several common attributes including a variety of elements coming from cultural associations (e.g. toponyms, artefacts, etc.), environmental characteristics (e.g. habitat types, land use), and physical domains (e.g. slope, altitude). These elements combine to distinguish a given place from others. If we refer to a landscape as a mosaic of places, boundaries can be defined as those zones “*of transition, contact, or separation between the contrasting elements of a mosaic*” (Cadenasso et al., 2003a).

In ecology, boundaries are characterized by two main components: (i) their structure gradual or abrupt changes in variables which define the transition (e.g. altitudinal changes, or administrative line partitioning), and (ii) their function — the flow of resources between adjacent patches, which might affect their composition (e.g. an increase in altitude which reduces temperature affecting vegetation structure and composition in a given place) (Strayer et al., 2003; Cadenasso et al., 2003b; Laurance et al., 2001). In Sacred Natural Sites (SNS) boundaries between places are determined not only by physical and ecological landmarks, but also by human perception (D'Eon and Glenn, 2000; Van de Noort and O'Sullivan, 2007), often shaped by secular religious taboos and spiritual norms. The praxis of combining physical features with mutable cognitive elements and values is common in such systems. Natural, physical, historical, administrative, and social elements contribute together to define the cognitive map, which people and communities use to delineate and distinguish what is sacred from what is not. Additionally, in SNSs and cultural landscapes, places are often referenced

¹The analysis and results of this chapter have been presented in one scientific conference (XIV World Forestry Congress, Durban, South Africa, 2015: peer-reviewed conference proceedings; Marini Govigli et al., 2015).

through the use of discrete toponyms, which work as a geographical reference system used by local communities to navigate their proximate surroundings (Alderman, 2008).

Boundaries of SNSs can be unfolded using the approach developed by Cadenasso et al. (2003b). Following this line of reasoning, we can decouple them into the perceptual boundaries' structure, i.e. the emotional gradient generating discontinuities (how we discern the environment itself, i.e. through fear, respect, etc.) and their function, which correspond to the difference in behaviour an individual would assume when crossing perceptual boundaries (e.g. if we access to such environment with augmented caution, fear, respect, etc.). By isolating boundaries' structure and function in SNSs as sacred forests, we can attempt at understanding how they have generated, and whether the boundaries still delineate active beliefs and practices. This is complex endeavour, due to the tight interactions amongst ecological, social, and cultural processes. As sacred forests change in time, due to changing social processes and direct and indirect ecological responses (Chapter IV), their boundaries might reflect this behaviour. Interactions that create boundaries are dynamic, in both space and time (Wild and McLeod, 2008). It is thus important to understand how boundaries of sacred forests are spatially determined and how their functions evolve as time passes. This can shed light on the processes that people used to determine the boundaries of their sacred forests.

To evaluate the interaction between social process, physical features and ecological responses, we sampled three sacred forests in NW Greece. The application of a multidisciplinary approach allowed us to overlay the groves' geophysical attributes (e.g. streams, slope) and ecological features (extent of large trees), as well as administrative infrastructure (e.g. roads, fences) and cultural prohibitions (e.g. grazing restrictions). This innovative approach helped to better understand the boundaries and elucidate the role of people's beliefs in boundary characterization. Lastly, it allowed us to verify if the spatial extent of sacred forests is determined by former management practices together with cultural perceptions and ecological drivers, proposing a new methodology to map

existing sources of knowledge related to religious and secular practices within a sacred landscape.

V.2 Materials and Methods

Three sacred forests were selected for this study. These are: (i) *Agios Nikolaos* in Livadakia, belonging to the village of Vitsa (ii) *Kouri*, village of Mazi; (iii) *Agia Paraskevi*, village of Vovoussa; more information on the forests is presented in Chapter II. These sites were selected for this detailed boundary analysis because of their similar structure, which gives us the best chance of distinguishing cultural patterns in boundaries' formation. All forests (i) surround outlying churches, (ii) have a clear comparable relationship with topography (located on tops of hills) and (iii), are small in size, allowing to sample the full extent of their areas.

In order to investigate the structure of the forests' socio-ecological boundaries, a specific framework was followed to identify all ecological, geophysical, cultural, and administrative elements, which might have contributed to sacred forests' boundaries formation. This involved:

- *Ecological boundary mapping.* Data collection of the main features believed to represent the ecological boundaries of the sites (changes in vegetation cover, spatial arrangement of trees and their corresponding ages);
- *Geophysical boundary mapping.* Topographical mapping including GPS recording of existing geophysical elements;
- *Cultural boundary mapping.* Identification of main perceptive elements associated with the forest by local communities (cognitive maps, toponyms), via interviews and secondary data sources;
- *Access infrastructure mapping.* GPS recording of main access routes to the forests (e.g. roads, paths, fences) via field visits.

Each identified boundary element was recorded and classified according to the definition presented by Cadenasso et al. (2003b), and the further elaboration of Strayer et al. (2003). All elements were assessed to elucidate: (i) their origin (where and when the boundary element was established), (ii) their spatial structure (e.g. their size, extent, thickness, dimensionality, etc.), (iii) their function (what was the reason behind the existence of that specific element characterizing boundaries), (iv), evidence of changes over time. The categorization of factors that can play a role in boundary definition adapted from Strayer et al. (2003) is presented in Table V.1.

V.2.1 Ecological boundary mapping

Linear boundaries of the forests as defined by Tsiakiris et al. (2013) were taken as a starting point for the study in that they defined the extent of area, which was investigated in detail. Tsiakiris et al. (2013) generates sacred forests' boundaries from ortho-rectified aerial photographs taken in 1945, when the forests were clearly visible as isolated patches in an intensively used landscape. Mid-20th century was identified as a period characterised by growing rural abandonment and human population decline, drastically associated with increasing tree infilling around the forests edges (Chapter IV). Nonetheless, 1945 aerial photos are the only available historical landscape snapshots of the forests, and, although abandonment was already occurring, they clearly show a demarcation between the sacred forests and their surrounding (treeless areas). We thus assumed these photos as the best available proxy to identify initial working boundaries for the study areas. Even so, the tree sampling scheme was not constrained by these limits and extended 50 m beyond them.

A two-stage tree inventory was undertaken in the three selected forests between June and September 2014 and 2015, using a systematic sampling followed by Adaptive Cluster Sampling based on the presence of large trees in the systematic plots (ACS; Thompson, 2012, 1990). ACS is an adaptive sampling scheme in the sense that the inclusion of new sampling units depends directly by the presence of an observed value, while conducting

Table V.1: Classification of boundary elements according to their origin and maintenance, spatial structure, function, and temporal dynamics. The classification system has been adapted from Strayer et al. (2003)'s ecological boundaries organization.

Boundary origin: How did the boundary originate, and how is it maintained?			
Investigative Boundary originated from mental human constructs		Tangible Boundary originated from an obvious and existing physical discontinuity	
Causal Boundaries originated because of existing patch discontinuities		Consequential Boundaries cause patch discontinuities	
Contemporary Boundaries originated from force still in operation		Relict Boundaries forces ceased	
Endogenous origin Generated within the study system		Exogenous origin Generated out of the study system	
Endogenous control Maintained by the study system		Exogenous control Maintained out of the study system	
Boundary structure: What is the spatial structure of the boundary?			
Dimensionality Dimensionality of the boundary element. Adimensional; 2D line; 3D spatial feature; Gradient; Fractal			
Type of change Type of change in conditions across the boundary element. Abrupt; Gradual			
Change extent Dimension of the change across the boundary element. Large change; Small change			
Integrity Level of integrity of the boundary elements. Unbroken; perforated.			
Boundary function: What are the functions of the boundary?			
Transformation The boundary element transforms the phenomenon	Amplification The boundary element amplifies the phenomenon	Reflection The boundary element reflects the phenomenon back	Neutral No effect on the phenomenon under study
Temporal dynamics: How does the boundary change over time?			
Changes in time Boundary element are still present or are fading in time. Present; Fading			
Mobility Boundary element are changing in space. Stationary (no changes); Changing (changes observed)			
Age and history Age of the boundary elements. Recent; Ancient			

the first survey (Thompson, 1990). If neighbouring sampling units satisfy the sampling condition, (i.e. they include the established observed value) they will be added to the original sample, and the process will continue until no further sampling units satisfy the original condition. ACS is a widely used technique in ecology for sampling both animal and plant populations (e.g. flora: Shackleton et al., 2020; Ojiambo and Scherm, 2010; Philippi, 2005; fauna: Cassel et al., 2020; Noon et al., 2006). In particular, it is used to assess ecological populations that tend to aggregate in clusters and are sufficiently rare in the environment (Pham et al., 2020; de Souza et al., 2020).

In this study, the rationale for using ACS was to facilitate the identification of clusters of trees, testing the working hypothesis that ancient protected trees tend to be clustered around the core of the grove (e.g. centre, church or icon-stand). The systematic sampling was designed with a square grid system (cell size 15m x 15m) with spacing varying accordingly to the forest areas (for additional details the reader can refer to Chapter II). In all plots, the main tree variables recorded were tree species and tree diameter at breast height (1.3 m). Further sampling using ACS was triggered by the presence of trees with diameter at 1.3 m (d) $\geq$ 40 cm, or stumps with basal diameter (d_0) $\geq$ 50 cm in the initial systematic plots in the oak forests of Mazi and Vitsa. In the pine sacred forest of Vovoussa, as pines are generally larger trees, the threshold to trigger ACS was taken as presence of trees with (d) $\geq$ 50 cm. These tree sizes were used as ACS triggering factors as they allowed fulfilling the condition listed in Christman and Lan (2001), namely allowing to have a proportion of tree individuals in the initial sample sufficiently rare (i.e. with a proportion of nonzero values very small, but not null). Following Thompson (1990), for each initial plot containing at least a tree over the specified threshold size criterion, a second set of neighbouring plots was laid out adjacent to each edge, and their trees sampled. This process was continued until none of the new plots satisfied the target condition (Table V.2). GPS coordinates for all trees sampled were recorded, along with species and diameter at breast height (1.3 m).

All sampled trees of the main canopy species (Deciduous *Quercus spp.* in Vitsa and Mazi,

Table V.2: Adaptive Sampling Cluster variables for the three study areas.

Sacred forest	Triggering condition for ACS	Number of initial systematic plots triggering condition (% on the total)	Number of second stage plots	Total number of plots (total number of tree sampled)	Area sampled (in ha) initial plots (total plots) ¹
Vitsa	Presence of at least a tree with diameter at 1.3 m $\geq$ 40 cm, or stumps with basal diameter $\geq$ 50 cm	7 (24.1%)	100	107 (239)	0.7 (2.4)
Vovoussa	Presence of at least a tree with diameter at 1.3 m $\geq$ 50 cm, or stumps with basal diameter $\geq$ 50 cm	13 (43.3%)	179	192 (423)	0.7 (4.3)
Mazi	Presence of at least a tree with diameter at 1.3 m $\geq$ 40 cm, or stumps with basal diameter $\geq$ 50 cm	19 (65.5%)	358	377 (764)	0.7 (8.5)

¹ Each sampled plot is 15 m x 15 m. More details in Chapter 2.

Pinus Nigra in Vovoussa), excluding dead individuals, were aged using age-diameter at breast height relationships derived in Chapter IV. The distribution of all aged trees was interpolated into a smooth surface of four main age classes (1700-1750, 1750-1800, 1800-1850, 1850-1915; three in Vovoussa) using Ordinary Kriging (Spatial Analyst toolbox, ESRI, 2020). Given that the estimated age of the minimum sampled tree size (diameter at breast height equal to 40 cm; 50 cm in Vovoussa), was of 110, 126, 141 years respectively for Vitsa, Vovoussa, and Mazi, a common upper bound equals to year 1915 was used in the kriging analysis. Kriging was run with a spherical semivariogram model, and a fixed searching radius for interpolation of 20 (Vitsa) and 30 (Vovoussa and Mazi) meters, changing according to the forest size.

V.2.2 Geophysical boundary mapping

In order to obtain topographical data at fine level for the three sacred forests, GPS surveys of altitude were conducted in the three forests on a 6 m grid giving 800

altitudinal points approximately per each forest. These survey data were integrated with the available ASTER global DEM (30m spacing) and used to interpolate a local DEM for each of the three sites. DEM rasters were employed to generate altitudinal maps and slope surface for each forests. Additional prominent geophysical features of the landscape were also mapped and georeferenced. A national layer of rivers and streams was used to interpolate existing watercourses (Geodata.gov.gr, 2015).

V.2.3 Cultural and administrative boundary mapping

The social data available was gathered from field visits, interviews with inhabitants of Vitsa, Mazi and Vovossa and a review of literature related to local history. Specifically, two interviews with agropastoralists who graze in Vitsa and in Mazi sacred forest were undertaken in 2014 (reported in Pion, 2014). One interview with a local inhabitant in Vovoussa was undertaken in 2015. Several field visits were done during 2014-2015, with experts and local inhabitants (e.g. as reported in Tsiakiris et al., 2019). Additional information was integrated from K. Stara personal ethnographical archive (Stara, *pers. comm.*) which was retrieved through extensive archival (manuscripts and monographs on local history and folklore) and ethnographical research in the region (Stara et al., 2016). Prominent social elements of the landscape (e.g. icon stands, churches, fences, paths, and roads) were also mapped and georeferenced.

Georeferenced data on the four data types (i.e. ecological, geophysical, cultural, and administrative boundaries) was entered into the ArcGIS (ESRI 2019) spatial software to produce land use maps for the three study areas.

V.3 Results

In the following section we first present the results of the boundaries classification, assessed according to their origin and maintenance, spatial structure, function, and temporal dynamics. Then the boundary elements associated with each forests are presented, alongside with a spatial visualization over the forest extent.

V.3.1 Boundaries elements of sacred forests

Table V.3 summarizes the 18 main ecological, physical and social elements characterising boundaries that were mapped across the three sampled sacred forests. Most of the elements are cultural (53%), 21% are administrative, while 16% and 10% are geophysical and ecological respectively. Concerning the boundary origins, data shows that most of the elements are tangible rather than investigative (74%), consequential rather than causal (68%), contemporary (74%) and of endogenous origins (74%). This indicates that boundaries demarcating sacred forest correspond to existing discontinuities in space, in many cases overlapping one with the other (e.g. slope lines aligned with roads, respectively marking separation between different management regimes). Most of them were established by users of the study systems, mainly by the local communities living in the surrounding of the forests.

Regarding their structure, 47% of the elements represent 3D spatial artefacts, and mostly indicate a gradual change of function rather than abrupt (53%), indicating that the associated change of behaviour, from a sacred area to its outer limits occurred bit by bit. Observing the boundaries functions, it emerges that 74% of the elements induces a transformation while only 37% an amplification, and 11% a reflection. This indicates that all elements identified generate some sort of behavioural changes in the forests users, the majority of which are related to transforming the behaviour of the trespasser (more cautious, respectful when inside of the sacred area). Lastly, 68% and 58% of the elements are assessed as still present in the system and as being stationary, respectively. Almost three fourths of the elements identified have an ancient origin, i.e. dating back to when the sacred forests were established (from the 16th to the 19th century approximately).

Table V.3: Summary of the main cultural, administrative, geophysical, and ecological boundaries elements identified in three sacred forests. Classification as in Strayer et al. (2003) presented in Table V.1.

Type	Name	Description	Boundary element origin	Boundary element structure	Boundary element function (sacredness of the area)	Temporal dimension	Present?
Cultural	Icon stands - Iconostasis	A small stand with a religious icon representing a saint or a holy event	Tangible, consequential, relict (partial), endogenous origin and control	3D element representing a bidimensional line, gradual change, large change in condition, perforated.	<u>Transformation</u> : Separation of the sacred forest from the outer zone, indicating an abrupt change in people's behaviour	Fading, stationary, Ancient	-
Cultural	Church	A church symbolizing the area of the forest over which the presence of the Virgin or Saint subjugates the forces of nature	Tangible, causal/consequential, contemporary, endogenous origin and control	3D element representing a spatial feature, gradual change, large change in condition, n.a,	<u>Transformation</u> : Separation of the sacred forest from the outer zone, implying a change in people behavior. It is believed that the church indicates the area with the strongest spiritual presence, the core of the sacred forest	Present, stationary, Ancient	Mazi Vitsa Vovoussa
Cultural	Management regimes	Management regimes (i.e. grazing, deadwood collection) occurring or prohibited withinsacred forests	Investigative, causal, relict, endogenous origin and control	Adimensional, abrupt change, large change in condition, perforated.	<u>Transformation</u> : Separation of the sacred forest from the outer zone, implying a change in people behaviour.	Fading, changing with time, Ancient	Mazi Vitsa Vovoussa
Cultural	Toponyms	Toponyms are used to colloquially refer to places within the forests	Investigative, consequential, relict, endogenous origin and control	Adimensional, gradual change, large change in condition, n.a,	<u>Amplification</u> : The usage of toponyms indicate a direct connection of people with their surrounding environment, allowing to distinguish the sacred from the non-sacred	Fading, changing, Recent/ Ancient	Mazi Vitsa Vovoussa
Cultural	Cross carved on trees	Cross were often carved on standing trees near the entrance of the forest or nearby the church	Tangible, consequential, contemporary, endogenous origin and control	3D element representing a spatial feature, gradual change, large change in condition, unbroken.	<u>Amplification</u> : Increase awareness of users that they are in a sacred site	Present, changing, Ancient	Mazi Vitsa Vovoussa
Cultural	Mental modes and beliefs	Individual beliefs of members of the community	Investigative, consequential, relict, endogenous origin and control	Adimensional, abrupt change, large change in condition, perforated-	<u>Transformation/Amplification</u> : The different beliefs of the community members towards the sacredness of the place implies a change of behaviour.	Fading, changing, Ancient	Mazi Vovoussa
Cultural	Ceremonies and knowledge transfer	Traditional ceremonies which take place within the sacred forests, and the extent to which they are transferred to new generations	Investigative, consequential, relict/contemporary, endogenous origin and control	Adimensional, n.a, large change in condition, n.a.	<u>Transformation/Amplification</u> : The different beliefs of the community members towards the sacredness of the place implies a change of behaviour.	Present/fading, changing with time, Ancient	Mazi Vitsa Vovoussa

Cultural	Tree shapes	Presence of pollarded, coppiced, shredded trees (deciduous trees), or trees tapped for resin (conifers). Presence of anthropogenic cuts (e.g. stumps)	Tangible, consequential, relict/contemporary endogenous origin and control	3D element representing a spatial feature, gradual change, large change in condition, perforated.	<u>Transformation</u> : The presence of worked trees indicates stronger interaction with the communities thus the out zones of a sacred forests	Fading, changing with time, Ancient	Mazi Vitsa Vovoussa
Cultural	Information signs	Signs which inform users on certain features of the sacred forests (vegetation, history)	Tangible, causal, contemporary, endogenous origin and control	3D element representing a spatial feature, abrupt change, small change in condition, n.a.	<u>Amplification</u> : Increase awareness of users that they are in a sacred site	Present, stationary, Recent	Vitsa Vovoussa
Admin.	Fences	A fence delimitating the extent of the forest	Tangible, consequential, contemporary, endogenous origin and control	3D element representing a bidimensional line, abrupt change, large change in condition, unbroken.	<u>Transformation/Reflection</u> : Separation of the sacred forest from the outer zone with a solid barrier, implying a change in people behaviour	Present, stationary, Recent	Mazi
Admin.	Roads And parking zones	Roads and parking for people to access the forest and its church	Tangible, consequential, contemporary, endogenous origin and exogenous control	3D element representing a bidimensional line, abrupt change, large change in condition, unbroken.	<u>Transformation/Amplification</u> : Roads give access to the sacred forest from the outer zone, implying a change in people behaviour. Parking might indicate the vicinity of the Church, increasing awareness of users that they are in a sacred site. In certain cases roads, coincidentally aligned with topography, indicate the limit of the forest extent	Present, stationary, Recent	Mazi Vitsa Vovoussa
Admin.	Gates	Gates to open the entrance to the sacred forest (set into tangible fence)	Tangible, consequential, contemporary, endogenous origin and control	3D element representing a spatial feature, abrupt change, large change in condition, n.a.	<u>Transformation/Reflection</u> : Separation of the sacred forest from the outer zone with a solid (although porous, as it can be opened) barrier , implying a change in people behaviour.	Present, stationary, Recent	Mazi
Admin.	Authorities demarking lines	Broader regional governance of natural areas can superimpose or negate boundaries of the sacred forests	Investigative, consequential, contemporary, exogenous origin and control	2D element representing a bidimensional line, abrupt change, large change in condition, unbroken.	<u>Transformation</u> : authority demarking plans can change they way people have to interact with the forest, by laws (e.g. prohibiting grazing)	Present, changing, Recent	Mazi
Geophysical	Slope	Surface inclination	Tangible, causal, contemporary, exogenous origin and control	Geo-physical gradient, gradual change, large change in condition, n.a.	<u>Transformation</u> : Slope and aspect can control vegetation cover with abrupt breaks in slope	Present, stationary, Ancient	Mazi Vitsa Vovoussa

					associated with both vegetation change and forest boundaries		
Geophysical	Streams	Small streams running near the forest	Tangible, causal, contemporary, exogenous origin and control	3D element representing a spatial feature, abrupt change, large change in condition, unbroken.	<u>Transformation</u> : Separation of the sacred forest from the outer zone, implying a change in land use	Present, stationary, Ancient	Vovoussa
Geophysical	Altitude	The vertical elevation of the surface	Tangible, causal, contemporary, exogenous origin and control	Geo-physical gradient, gradual change, large change in condition, n.a,	<u>Transformation</u> : The surface elevation can be correlated to different vegetation cover, thus associated to the boundary extents	Present, stationary, Ancient	Mazi Vitsa Vovoussa
Ecological	Tree ages	Trees within the sacred forests can be clustered by age classes	Tangible, consequential, coeval, exogenous/ endogenous origin and control	3D element representing a spatial feature, gradual change, large change in condition, perforated.	<u>Transformation</u> : Veteran trees characterize the main core of the sacred forests. The oldest cohort of trees is often located in the proximity of the church	Present/fading, stationary, Ancient	Mazi Vitsa Vovoussa
Ecological/ cultural	Vegetation cover changes	Zones of transitions between two adjacent habitats.	Tangible, causal/ consequential, relict/ contemporary, endogenous origin and control	3D element representing a spatial feature, gradual change, large change in condition, perforated.	<u>Transformation</u> : changes in vegetation can indicate the limit of the sacred forest. They can occur naturally or artificially	Present/fading, changing with time, Ancient	Mazi Vitsa Vovoussa

V.3.2 Overlaying boundary elements in land-use maps

Figure V.1 shows a map of Vitsa with its associated boundary elements emerging from Table V.3. The map includes four boundary elements indicating the extents of the core of the sacred forest which is taken as including: (i) the church, (ii) the presence of crosses carved on trees, (iii) the presence of the oldest canopy trees, (iv) the hill tops (highest altitudes), and the absence of three elements that generally indicate areas outside the sacred forest: (i) roads, (ii) observed anthropogenic cuts (branches sawn and stumps), (iii) and the occurrence of active management (grazing). The census of the largest trees in Vitsa arising from the ACS includes 239 individuals, the majority of which are deciduous oaks of four main age classes: an ancient cohort of 6 individuals established between years 1700-1750 a cohort of 15 individuals established between years 1750-1800, a medium-aged 39 trees dating back to 1800-1850, and 157 individuals established between 1850 and 1915. The dominant species are *Quercus frainetto* (150 individuals) and *Q. cerris* (61 individuals). Tree distance to the church is negatively correlated with tree age (Pearson Correlation Coefficient: -0.27; $p < 0.01$), showing how the oldest individuals of more than 200 years old (established before 1800) are clustered close to the church of Agios Nikolaos (average distance from church = 77.5 ± 2.3 m). The oldest cohort of trees is also located within or alongside additional boundary elements associated with the inner zones in sacred forests, in this case the presence of trees with carved crosses. Conversely, areas with higher levels of current human interference in term of grazing and anthropogenic cuts are mostly located further away from the church and core of the forest.

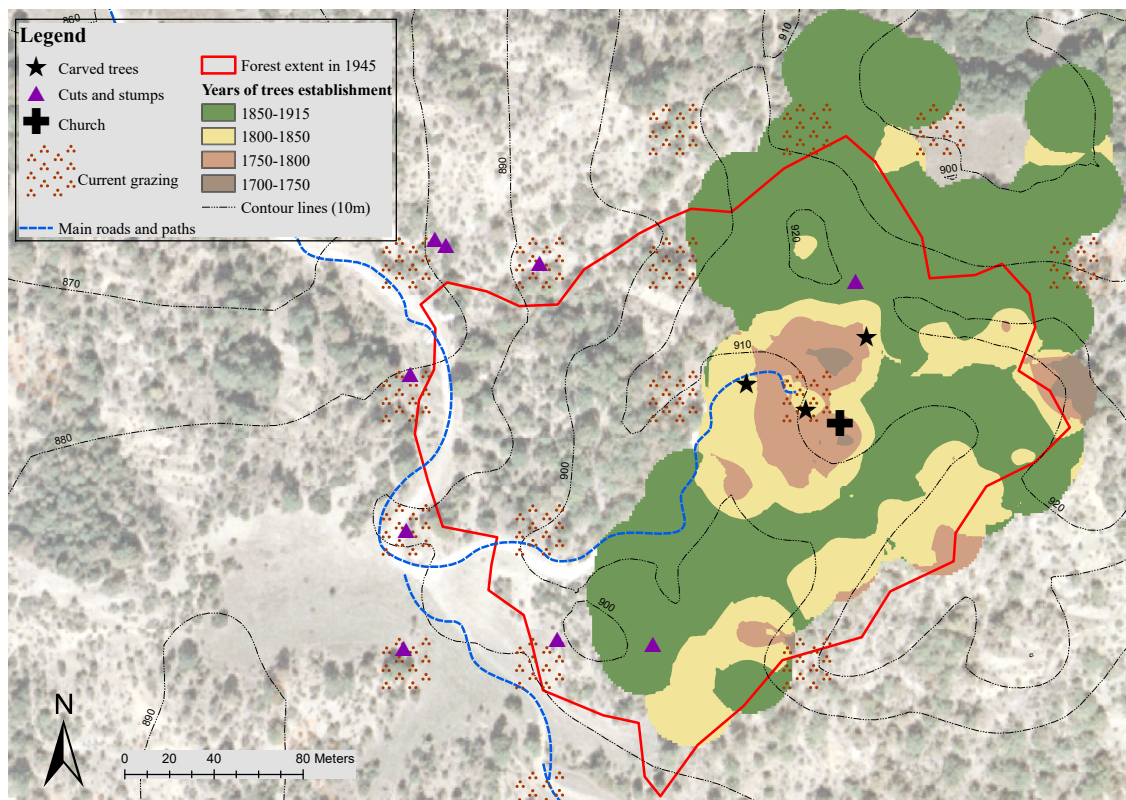


Figure V.1: Extent and structure of Vitsa sacred forest overlaid on its satellite image (2007). The map shows the presence of eight main boundary elements: the church, the main road to the forest, the presence of cross carved on trees, the extent of the oldest cohorts of canopy trees, the altitudinal gradient, presence of anthropogenic cuts and occurrence of active grazing. The coordinates of the aged trees (*Quercus frainetto*, *Q. cerris*, *Q. pubescens*) were interpolated into a smooth surface of trees age coloured to represent four age classes using kriging. The red line indicates the extent of the forest in 1945 as estimated from aerial imagery after Tsiakiris et al. (2013).

In Vovoussa ten elements were mapped (Fig. V.2). Five elements representing the core of the sacred forest: (i) the church and (ii) its main path, (iii) presence of cross carved on trees, (iv) the location of the oldest canopy trees, and (v) the altitudinal gradient; and five elements representing areas outside the forest: (i) the occurrence of management (former agricultural fields), (ii) worked trees (presence of sawn branches and resin-mined pines), (iii) streams and (iv) roads. The ACS survey recorded 423 trees greater than 50 cm in Vovoussa. The majority of *P. nigra* trees fell into three age classes: an ancient cohort of 6 individuals established between years 1750-1800 a cohort of 55 individuals established between year 1800-1850, and 299 individuals established between 1850 and 1900. Only

26 large trees were of species other than pine these being *Quercus spp.*, *Carpinus spp.*, and *Acer spp.* for which age was not inferred. The oldest individuals, established more than 200 years ago, are clustered close to the church of Agia Paraskevi (average distance: 109.6 ± 2.2 meters). As in Vitsa, distance to the church is negatively correlated with tree age (Pearson Correlation Coefficient: -0.31 ; $p < 0.01$). The oldest cohort of trees is also located alongside features associated with inner zones in sacred forests such as the presence of crosses carved on trees, and dead standing trees. On the other hand, former agricultural areas, and worked trees are mostly located away from the core of the sacred forest.

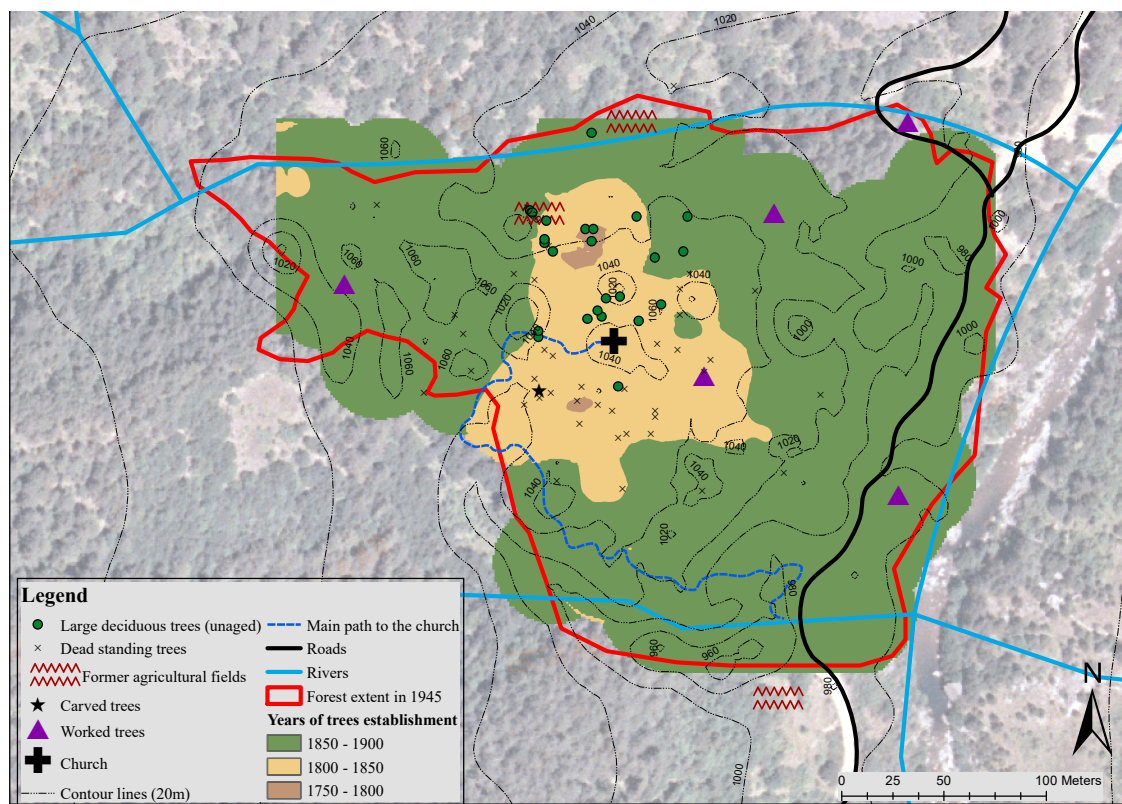


Figure V.2: Extent and structure of Vovoussa sacred forest overlaid on its satellite image (2007). The map shows the presence of nine boundary elements: the church, the main road and foot-path to the forest, the presence of cross carved on trees, the location of the oldest cohorts of canopy trees, the altitudinal gradient, worked trees (presence of sawn branches and resin-mined pines), and occurrence of former agricultural fields, and rivers. The coordinates of the aged trees (*Pinus nigra*) were interpolated into a smooth surface of three age classes using kriging. The red line indicates the extent of the forest in 1945 as estimated from aerial imagery after Tsiakiris et al. (2013). A national layer of rivers and streams (light blue line) was used to interpolate existing watercourses. Due to the coarser precision level of this layer, its exact spatial extent has to be carefully interpreted.

The land-use map of Mazi sacred forest shows the interplays of ten boundary elements (Fig. V.3). Five of these elements represent the core of the forests (churches, cross carved on trees, the ecological extension of the oldest cohorts of canopy trees, the forest altitudinal gradient), while the remaining ones represent its outer limits (the extension of the fence and gate enclosing the forest, the occurrence of active management, presence of working trees, and roads extent). There are 764 large trees (diameter at breast height > 40 cm) in Mazi sacred forest, the majority of which are *Quercus*. Five main age classes of oaks have been identified: an ancient cohort of 5 individuals established between years 1650-1700; with one individual estimated to date of around early 16th century; a cohort of 16 individuals established between years 1700-1750; a cohort of 70 individuals established between years 1750-1800, a cohort of 232 trees dating back to 1800-1850, and 330 individuals established between 1850 and 1900. Dominant species are *Q. pubescens* (255 individuals) and *Q. trojana* (231 individuals). Only 41 sampled trees belong to species other than oaks these being *Phillyrea latifolia*, *Fraxinus ornus*, and *Ulmus minor* whose age could not be inferred. Unlike Mazi and Vouvoussa, no correlation was found between the distance to either of the two churches and the oldest trees, as the oldest individuals were mostly located near the current entrance of the forest (the eastern gate). In this location are located a cluster of other features usually associated with inner zones in sacred forests, such as the presence of crosses carved on trees. Whereas areas with higher level of current human use in term of grazing and anthropogenic cuts are mostly located away from the oldest trees and nearby the churches, differently from what found in Vitsa and Vovoussa.

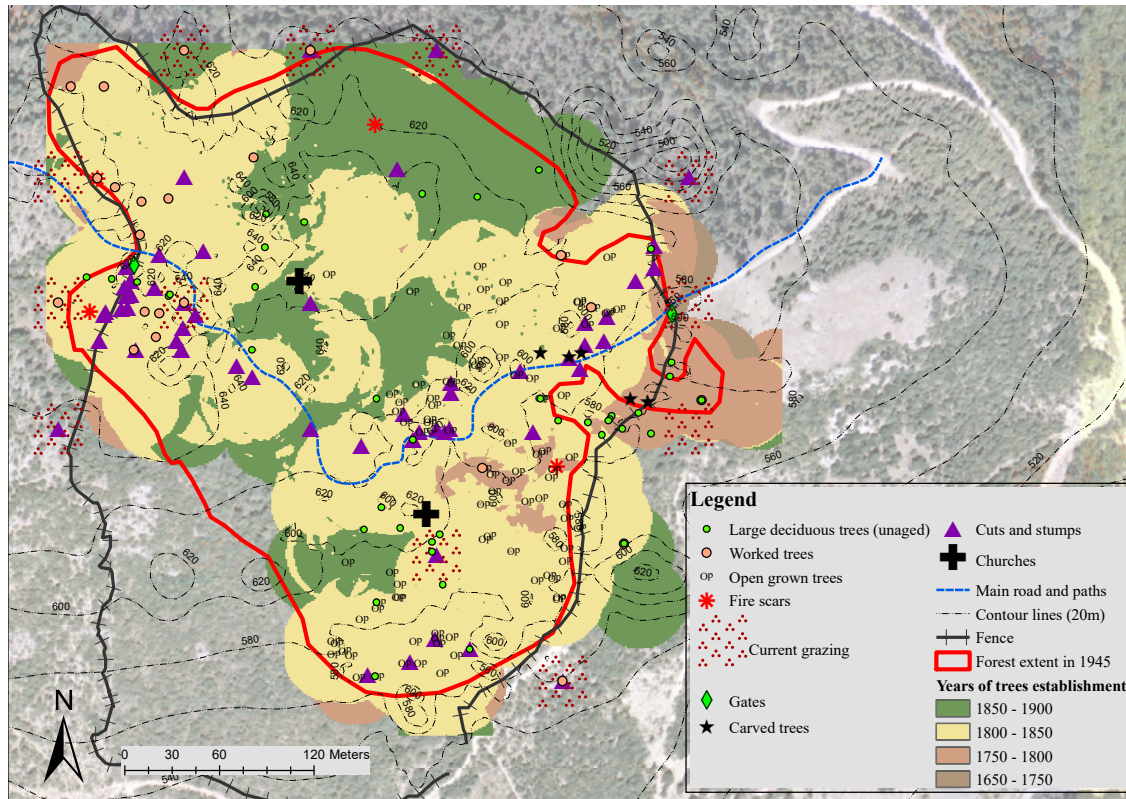


Figure V.3: Map of Mazi sacred forest overlaid on a terrain satellite image (2007). The map shows the presence of nine boundary elements: the churches, the main road to the forest, the presence of cross carved on trees, the ecological extension of the oldest cohorts of canopy trees, the altitudinal gradient, anthropogenic cuts, worked trees (pollards, shreds, and coppices), occurrence of current management (grazing), and the extension of the fence and gate enclosing the forest. The coordinates of the aged trees (*Quercus frainetto*, *Q. cerris*, *Q. pubescens*, *Q. trojana*) were interpolated into a smooth surface of four age classes using kriging. The red line indicates the extent of the forest in 1945 as estimated from aerial imagery after Tsiakiris et al. (2013).

V.4 Discussion

Our analysis allowed the identification of the main boundary elements that characterize the transition between core and peripheral areas of sacred forests in Northern Greece. This is a paramount exercise for SNSs, as it allows shedding light on the process that local communities used to determine the boundaries of their sacred forests, having direct implications for their current conservation and management. Additionally, this analysis helps to understand whether we can demonstrate that the reciprocal interplay of natural, cultural, perceptive, and geophysical factors has a pivotal role in defining the

spatial configuration of Northern Greece sacred forests. In this section, we further explore these findings to identify spatio-temporal patterns in the internal structure and boundary conditions across the three case study forests.

V.4.1 Spatial dynamics

The analysis performed in Vitsa, Vovoussa and Mazi shown three main elements that have emerged as key for shaping the forests' boundaries: (i) the location of the churches on hill tops (which was expected having the three sites been selected to have central churches on hilltops), (ii) the clustering of the oldest cohorts of canopy trees around them, and (iii) the location of managed trees and landscape areas in contraposition to the prohibition regimes established within the sacred forests.

Two out of the three sampled forests (Vitsa and Vovoussa) have the oldest trees (dating from the 16th-17th century) located in proximity to a central church. This is typical of many existing sacred sites in which the church, together with icon stands or other religious structures, are often identified as the cultural centre of the site itself (Lagopoulos, 2002; Aerts et al., 2016). All three sampled sacred forests are dedicated to Orthodox saints and currently used for local celebrations in situ: Vitsa to *Agios Nikolaos* (Saint Nicolas 20th of May), Vovoussa to *Agia Paraskevi* (26th of July), while Mazi to both *Prophétes Elías* (Prophet Elijah 20th of July), and *Evangelístria* (Virgin Mary 25th of March). In pastoralist communities such as Mazi and Vitsa, these festivities were associated with the starting of the grazing season. In Vovoussa, the feast of Agia Paraskevi brings the whole community to the forest to celebrate the Saint with a religious service and a procession with songs. This traditional ritual still occurs nowadays (Fig. V.4). The central role of the church in the studied sacred forests, suggest that churches might act as the locus causing discontinuities between the portion of the forest that can be inferred as sacred and the one that is not. Local communities would then treat the environment surrounding the church differently, as symbolizing the area over which the presence of the Virgin or Saint was stronger. Such perceptual constraints could hence have acted as

a powerful spiritual boundary which may have contributed to generate the current extent of the sacred forests (Stara et al., 2015a; Nixon, 2006).



Figure V.4: Saints' celebrations in sacred forests. From left to right: the celebration of Agios Nikolaos in Vitsa sacred forest; the sung procession down the forest path for the day of Agia Paraskevi in Vovoussa. ©Marini Govigli V.

On the contrary, in Mazi sacred forest, the oldest cohort of trees is located nearby the Southern church of Evangelístria (dated 1755) and the current entrance of the forest, where additional religious symbols (i.e. the crosses carved on trees) and the village aquifer can be found. This suggests that the oldest core of the sacred forest is located elsewhere from where it is currently considered to be (the areas constrained by the two churches). This finding is consistent with available historical narratives, which says that the churches were originally the entrance of its village (which has since moved location down in the valley) and pointing out that one of the main services attributed to the sacred forest in the past was replenishing village aquifers (Tsiakiris et al., 2019).

It is however not yet clear if the sacred forests' configuration has been determined by spiritual and perceptual constraints or by topographical factors, as both being close interrelated. The forests topographical arrangement, i.e. distinctive hills in a generally flat environment (Vitsa) or close to a village (Mazi and Vovoussa), could have facilitated the community leaders to better define their former boundaries, and for the community

to respect it. Maps produced for Vitsa and Vouvoussa (Fig. V.1 and V.2) show clearly how the oldest cohorts of trees are located on the top of the hill and so have likely been part of the sacred forest from the beginning.

Lastly, a third dominant factor (beside topography and spiritual constraints) shaping the forest extents are former and current management practices within and outside the sacred forests. Vitsa and Mazi are sacred forests of agro-pastoralists communities (Chapter II). Prohibition of cutting trees was enforced within them but shepherds and goatherds were granted the use of the forests (after the celebration of the local Saint) for grazing and for resting their herds in the shade of the large oaks trees. This use is similarly documented in wood pastures elsewhere in Europe (i.e. Hartel et al., 2013; Bergmeier et al., 2010). In Mazi the former use of the area for grazing is evidenced by the presence of numerous large trees with horizontally spreading lower branches (Fig. V.5), suggesting that they have originated in a more open landscape (Marks and Gardescu, 2001). In Vovoussa, the church forest used for religious rituals is associated with strict access restrictions as deadwood removal and even hunting or the collection of wild honey were not allowed. The solidity of this cultural taboo was confirmed by the large amount of dead standing trees, which have been found in the proximity of the church. Trees subjected to shredding, pollarding, or coppicing for tree fodder are not generally found within Northern Greek sacred forests though these are common practices outside the sacred forests (Stara et al., 2015a). This is confirmed by our results showing that anthropogenic stumps, cuts, worked trees, and former agricultural areas are mostly located in the areas surrounding the forests, indicating the outskirts of the sacred space. In Mazi, the profuse presence of stumps within the current sacred forest limitations, indicate that tree cutting was prescribed at a certain moment in time (and precisely from 1975 to 1977 for collecting firewood, and in 1995 for wood selling and income generation; Tsiakiris et al., 2019).

Results from the boundary assessment (Table V.3) also indicate that most boundary

elements generate a gradual change of function, rather than abrupt changes. This is because sacred forests configuration, in many respects, depend on the way the different members of a community react to rules and moral codes (Stara et al., 2016). Thus, human perception on religious and traditional rules can differ greatly between different groups within a village (e.g. shepherds vs. summer residents; Efthimiou, 2019) and thus one group may observe a rule while another does not, resulting in a fuzzy and diffuse boundary. Geophysical features (i.e. rivers in Vovoussa), or unbroken administrative lines (i.e. fences in Mazi, Fig. V.5) are those mostly associated to abrupt change in function, which can cause sudden patch discontinuities. Sudden line partitioning was additionally identified as source of conflicts between different groups of users or stakeholders in the forest. In Mazi, the building of a fence enclosing the forest in 1982 by the ecclesiastical and community councils and the Forestry Service, created frictions with certain local families opposing this decision. Similarly, Forestry Service restrictions imposed in 1922 on grazing to reduce fire risks were challenged by those shepherds grazing within Mazi sacred forest (Tsiakiris et al., 2019). Similar conflicts over boundaries definition and resource management held in commons have been identified by literature in other regions of the world (Quinn et al., 2007; Johnson and Nelson, 2004).



Figure V.5: Elements characterizing boundaries in sacred forests. From left to right: a deciduous oak with former horizontally spreading branches in Mazi; the gate of the 1982 fence surrounding Mazi sacred forest; a cross carved on the bark of a deciduous oak. ©Marini Govigli V.

V.4.2 Temporal dynamics

The results of this study have shown that the centre of the three sacred forests date back to the 16th and 17th centuries. This conforms to available historical data on the villages which indicates that the sacred forests were established during the period of Ottoman occupation (1479- 1912; Lambridis, 1870; Chapter IV). Nonetheless, depopulation trends and rural abandonment have weakened the key role of sacred forests within the local communities (Stara et al., 2016). Our results show that many of the elements associated with boundary in northern Greece sacred forests are now fading. The disappearance of cultural elements has a direct effect on their traditional associated function, i.e. the transformation of peoples behaviour as they approach the core of the sacred forests (Table V.3). If changes of behaviour and knowledge of perceptual boundaries are lost, then the socio-cultural construct, which generated and maintained sacred forests through the centuries, is also at risk, especially when combined with drastic ecological changes i.e. occurrence of fires².

These changes are also worsened by the abandonment of former agricultural lands over the past 100 years (Papanastasis, 2007), which turned sacred forests and their surroundings into a continuous forest cover. Former open fields are colonized by tree species present in the neighbouring sacred forests. Forests are thus expanding out from sacred areas into parts that are not subjugated to religious forces, hence not traditionally bounded by any management prohibition (with the exception of Vovoussa sacred forest where a number of agricultural fields were donated to the forest property, Stara, 2016, *Pers. Comm.*). This is particularly relevant as it indicates how the ecological expansion of sacred forests does not necessarily imply an increase of the sacred areas, as their boundaries rely on both ecological elements and local cultural beliefs.

Lastly, temporal dynamics indicates how the perception of people is changing through

²Fire scars identified in Mazi suggest that a large wildfire might have occurred in the past probably contributing to modify the spatial extent of the forest (Tsiakiris, 2016, *Pers. Comm.*).

time. The presence of recent information boards in some of the sampled sites shows how religious devotion is evolving into environmental awareness and heritage preservation. Pro-environmental trends, including land stewardship, are indeed emerging values of many Western Christian traditions (Hope and Jones, 2014; Greeley, 1993). On the other hand, cultural abandonment and loss of tradition are associated with loss of consciousness of the sacred sites existence. Understanding the evolution of peoples' perceptions and the effects it can have on the sacred forest condition and boundaries is thus required for the development of effective management strategies of Sacred Natural Sites.

V.5 Conclusion

The main purpose of this study was to assess how boundary elements of sacred forests are determined, and whether they depend on the interplay of different natural, cultural, perceptive, and geophysical factors. We conclude that: (i) the current boundary configuration of northwestern Greek sacred forests is the result of a set of diverse socio-ecological processes that, combined, contributed to influence its structure; (ii) such processes have a temporal component, with the oldest tree communities often located in the core of the sacred forests and the vicinity to the church or other significant feature such as water aquifers; (iii) anthropological aspects such as management regimes within and in the surrounding of the groves and perceptual constraints played an important role in the dynamics of the sacred forests boundaries.

We also have demonstrated that socio-ecological constructs as Northern Greece sacred forests are too fuzzy to be definable tracing a simple line, as the attempt followed in Tsiakiris et al. (2013) using 1945 aerial images. Their boundary elements are too diffuse and depend on a multiplicity of factors, which make them difficult to map. This important finding has direct implications for the conservation and future management of the sites as require practitioners to adopt multidisciplinary approaches to identify their spatial extent.

This research has successfully provided the basis to better understand boundary dynamics of fuzzy systems such as sacred forests. While additional case studies need to be investigated to further test the effectiveness of the methodology, available evidence highlights how boundary expansions in sacred sites can happen only with the combined interplay of ecological and social changes. Depopulation, loss of local tradition, and forest expansion can pose a severe challenge for the preservation of sacred forests as intangible local cultural heritage. Understanding boundaries plasticity in complex Socio-Ecological Systems become thus relevant for developing ad hoc conservation strategies.

CHAPTER VI

Modelling the socio-ecological development of sacred forests in Epirus using a Cellular Automaton model

VI.1 Introduction¹

Sacred Natural Sites are cultural-natural landscapes arisen from the interplay of social, ecological, physical and geomorphological processes across time (Verschuuren, 2010). They are dynamic constructs that directly depend on their driving forces. Therefore, if one (or more) of these fades or changes, chain effects can occur on connected processes affecting ultimately the overall landscape.

In long-term Socio-Ecological Systems as sacred forests, social and ecological processes are very likely to generate interdependences (Liu et al., 2007). Changes in demography, management, and traditional knowledge are directly interconnected with natural and ecological processes (i.e. infilling of secondary vegetation in abandoned fields) (De Aranzabal et al., 2008). Understanding the dynamic response of these changes both in time and space is thus crucial for appraising the overall evolution of the landscape matrix, and in turn to map dominant driving forces and assess whether different social and economic processes can effect simultaneously the vegetation structure and its diversity (Favier et al., 2004).

Vegetation models are useful tools that can be used by practitioners to identifying the above-mentioned recurring patterns in forest systems. In opposition to Clements' Climax theory, starting from the late 20th century, a series of novel vegetation modelling approaches began to be developed (Briske et al., 2005). These models reflected changes in Ecology generally, which saw a paradigm shift away from the Balance-of-Nature concept towards something more dynamic in which ecological systems are constantly in a state of flux (Halley, 1996; Pimm, 1991). Ecosystems are thus unstable structures whose species can fluctuate over time and space due to external shocks or internal dynamics. They are now seen as entities in flux on different timescales due to external

¹The analysis and results of this chapter have been presented in one scientific conference (5th meeting of the forest modelling working group, Solsona, Spain, 2019; oral presentation with title *Modelling a Socio-Ecological System. A Cellular Automata model for the spatial development of sacred forests in Epirus, Greece*).

and internal dynamics. This theoretical strand opened the door to transition models (Westoby et al., 1989). In these modelling interfaces, vegetation species move from one state to another according to given transition parameters (i.e. survival, mortality, and fecundity rates). These approaches, rather than a climax, focuses on the trajectory that a given ecosystem is most likely to reach in the future, given a set of pre-determined starting conditions.

Lately, thanks to modern computational advances, several of these models have been integrated as to simulate vegetation changes in space (i.e. Spatially Explicit Models; He, 2008). These models allows to map ecological processes at finer precision levels, assessing behavioural patterns at varying spatial and temporal scales (DeAngelis and Yurek, 2017). Many types of spatially explicit models exist, including: patch models, neighbourhoods models, individual and agent-based models, and cellular automata (for a review, refer to DeAngelis and Yurek, 2017). Cellular Automata (CA) are dynamic models explicit both in time and space determined by given number of spatially interconnected cells (Wolfram, 1983). CA are thus applied to many fields, including population dynamics of single species (Cannas et al., 1999; Bandini and Pavesi, 2002) competitive interactions across different plant species (Jie Xu et al., 2010; Balzter et al., 1998; Silvertown et al., 1992), invasive species management (Lu et al., 2013), landscape changes due to natural and anthropogenic disturbances (Obade et al., 2014; Onsted and Chowdhury, 2014; Favier et al., 2004), species extinction (Darwen and Green, 1996), fire dynamics (Krenn and Hergarten, 2009; Franciosi, 2009), socio-ecological systems (Engelen et al., 1995), and many more. In particular, in the field of plant ecology their applications has increased, due to the need of modelling plant population trajectories under complex spatio-temporal processes. These models also allow to better capture the intrinsic complexity of SESs, depicting changes in human behaviour more realistically, taking into consideration the mutual interactions of heterogeneous human decisions and choices over the landscape (Filatova et al., 2013). The use of CA to understand the historical evolution of sacred natural sites remains still an unexplored field.

The purpose of this research is to develop a CA interface able to assess the spatial extent of sacred forests over changing socio-ecological constraints. The main goal is determining to what degree socio-economic processes (i.e. demographic and management changes) can impact the spatial extent of sacred forests otherwise determined by ecological processes of forest expansion. To make a step further, we use in this study a $1/f$ -noise process as a model of environmental variability in space and time, to describe fluctuations in survival of seeds. Stochastic models with structured noises are being used in Ecology as representations of all those environmental influences which cannot otherwise be explicitly included in models. By using $1/f$ noise for modelling spatio-temporal variability, we are able to give equal weight to environmental shocks happening at different spatial and temporal scales, disentangling both common and rare disturbance drivers (Halley and Inchausti, 2004 and Section VI.2.2). We find this approach suitable for modelling something as complex as a "sacred forest system". Several studies have used spatio-temporal simulations for appraising socio-economic processes (e.g. for urban growth or land-use and land-cover changes; Aburas et al., 2017; Deng et al., 2004). To the best of our knowledge, this is the first time that a $1/f$ noise process is used to produce spatio-temporal simulations of a system under both socio-economic and ecological constraints.

This research builds on the findings of the former chapters in which we demonstrated that: (i) sacred forests main vegetation cycles are affected by the strength of large societal changes on the territory (Chapters III and IV), and (ii) that many of these processes might have a combined effect on the temporal evolution of the spatial configuration of sacred forests (Chapter V). In this section we further explore these findings by developing and implementing a novel theoretical approach that allows us to estimate the joint impacts of population flows, management changes, and strength of taboos and prohibition regimes on the extent of one sacred forest. By applying this model we corroborate oral and historical narratives, and build a tool flexible enough that can be applied to various cultural landscapes. Moreover, we make a substantial

contribution to the existing literature in ecological modelling, providing an unique application of a $1/f$ -noise process to model environmental variability in space and time; a step further towards parametrizing socio-ecological systems using spatio-temporal correlations in Ecology.

VI.2 Model development

VI.2.1 Model structure

A general Cellular Automaton (CA) is described as follows:

$$CA = (d, Q, \delta, f) \quad (1)$$

where d is the dimension of the lattice E in which the model is set (the cell grid); Q the state space of the cells in E ; δ the neighbouring template of the model, and f the local transition function. These are the basic arguments which determine the behaviour of the automaton in time (according to the state of the cells and the rules for their transition), and in space (according to the extent of the lattice and the cell neighbouring template). In this study we coded a CA model that predicts the spatial extent of a sacred forest-type landscape under specific socio-ecological constraints: MOSES (Model of a Spatially explicit Socio-Ecological System)². MOSES is structured as a CA interface whose cells, representing spatially explicit forest fragments, evolve in time and space under a combination of several governing rules and socio-environmental constraints.

Under these specifications, the next state (i.e. the cell condition) q of the i th cell at time $(t + 1)$ is then equal to:

$$q_i(t + 1) = f_i(q_i(t), Z_i(t), \delta_i) \quad (2)$$

where $q_i(t)$ denotes the state of the i th cell at time t (i.e. the tree population within the cell at time t); $Z_i(t)$ represents the social variables that further affect the vegetation structure of the i th cell at time t (e.g. prohibitions in place, management changes, etc.); f_i

²The model has been coded in Python 2.7 language (Python Software Foundation, 2018). Model codes are provided in Appendix N.

is the state function (the rule of the automata), and δ_i the neighbouring template of the i th cell. In MOSES cells change synchronously their states in discrete (yearly) time-steps. In particular, the neighbouring template is set as a Moore neighbourhood, *i.e.* including the eight neighbouring cells of the i th cell (Weisstein, 1999). The neighbouring template influences the tree population of the i th cell via seed dispersal rates (Fig. VI.1).

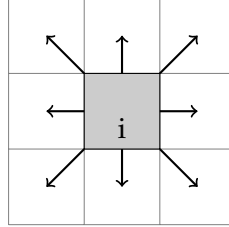


Figure VI.1: Close up of the MOSES neighbouring template, showing how seeds will be dispersed from the cell i to its eight neighbours, and vice versa.

VI.2.2 The ecological layer

Population structure of the cells

MOSES is structured as a single species model in which each cell i represents a forest plot of size 15x15 m characterized by a tree species n in all its life stages. Specifically n is a seven dimensions vector described as:

$$\mathbf{n} = [n_0, n_1, n_2, n_3, n_4, n_5, n_6] \quad (3)$$

in which each element represents a different stage of the tree life cycle in term of diameter size classes (Table VI.1). The evolution in time of the tree population of each cell is described by a Lefkovitch matrix. These approaches are often used in ecology to describe discrete and size-structured model of population growth and survival (e.g. as in Collevatti and Hay, 2011; Zhang et al., 2008; Valverde and Zavala-Hurtado, 2006). While Leslie matrices assess population changes in term of mortality and fecundity, Lefkovitch matrices replace age classes by ontogenetic³ stages of the studied organism

³“The development of an individual organism from its earliest stage to maturity”; Oxford Dictionary, 2020

(Lefkovitch, 1965). This structure is often used by practitioners when age parameters are hard to infer, as in the case of marine mammals (Carnia et al., 2015). Additionally, they are widely used in forest ecology, where uneven relationships can occur between tree age and size variability (Yusoff et al., 2012).

Table VI.1: The structure of the seven stages tree population vector. d_1 is the diameter of the tree measured at breast-height; d_0 is the diameter of the tree measured at ground level.

Vector element	Tree life stage	Size class in cm
n_0	Seeds	-
n_1	Seedlings	$d_0 < 1$
n_2	Saplings	$d_0 \geq 1$ AND $d_1 < 5$
n_3	Poles	$5 \leq d_1 < 10$
n_4	Adult trees 1	$10 \leq d_1 < 30$
n_5	Adult trees 2	$30 \leq d_1 < 50$
n_6	Mature trees	$d_1 \geq 50$

For each model cell, given the following rates:

- $g_j(t)$, the probability of tree individuals to survive and grow from size class j to size class $j + 1$ at time t ;
- $s_j(t)$, the probability of tree individuals to survive in class j and not to grow to size class $j + 1$ at time t ;
- $m_j(t)$, the probability of tree individuals to die in class j at time t ;
- $f_j(t)$, the fecundity as the average number of seeds produced by one individual of the size class j at time t ;

then the number of individuals n_j at time $t + 1$ of size $j + 1$ ($n_{(j+1)}(t + 1)$) is equal to the proportion of individuals at time t of size j that survived and grew into the next class ($j + 1$) plus the number of individuals at time t of size $j + 1$ that survived but did not grow to the next size class:

$$n_{(j+1)}(t + 1) = n_j(t)g_j(t) + n_{j+1}(t)s_{j+1}(t) \quad (4)$$

This with:

$$g_j(t) + s_j(t) + m_j(t) = 1 \quad (5)$$

Given n that goes from 0 to ω (the maximum size class of the population, i.e. equal to 6; Eq.3), in each i th cell of the CA the following Lefkovitch matrix stands:

$$\begin{bmatrix} n_0 \\ n_1 \\ \vdots \\ n_{\omega-1} \\ n_\omega \end{bmatrix}_{t+1} = \begin{bmatrix} s_0(t) & f_1 & \dots & f_{\omega-1} & f_\omega \\ g_0(t) & s_1 & \dots & 0 & 0 \\ 0 & g_1 & \ddots & \vdots & 0 \\ \vdots & \vdots & \ddots & s_{\omega-1} & \vdots \\ 0 & 0 & \dots & g_{\omega-1} & s_\omega \end{bmatrix} \begin{bmatrix} n_0 \\ n_1 \\ \vdots \\ n_{\omega-1} \\ n_\omega \end{bmatrix}_t \quad (6)$$

This can be written in vector form as:

$$n(t+1) = M(t) \times n(t) \quad (7)$$

where $n(t)$ is the tree population vector at time t and M denotes the Lefkovitch matrix. Time t influences the Lefkovitch matrix through parameters s_0 and g_0 as seed mortality is affected by spatio-temporal fluctuations (as presented in the following section).

Equation 6 stands in the case of a closed system (i.e. a system in which individuals are not allowed to immigrate or emigrate to neighbouring cells). In MOSES, tree individuals of dimension $n \geq 3$ will produce and disperse seeds to the neighbouring cells (an 8-cell Moore neighbourhood). Although seeds can disperse to large distances (Veresoglou and Halley, 2018), in this model we constrain trees to disperse seeds only to their 8 neighbouring cells as we focused our analysis on understanding the effects of selected socio-economic processes on the sacred forest immediate extent (hence short range dispersal) through time ⁴. The q state of the i th cell at time $t+1$ is then equal to:

$$\begin{bmatrix} n_0 \\ n_1 \\ \vdots \\ n_{\omega-1} \\ n_\omega \end{bmatrix}_{t+1} = \begin{bmatrix} s_0(t) & f_1 & \dots & f_{\omega-1} & f_\omega \\ g_0(t) & s_1 & \dots & 0 & 0 \\ 0 & g_1 & \ddots & \vdots & 0 \\ \vdots & \vdots & \ddots & s_{\omega-1} & \vdots \\ 0 & 0 & \dots & g_{\omega-1} & s_\omega \end{bmatrix} \begin{bmatrix} n_0 + (\mu\hat{n}_0 - \zeta n_0) \\ n_1 \\ \vdots \\ n_{\omega-1} \\ n_\omega \end{bmatrix}_t \quad (8)$$

⁴To include an estimation of the deposition of seeds spread by bird, wind, humans, and animals in the CA, we parametrized a seed rain wave, which each year, can deposit additional seeds n_0 in all system cells.

where ι is the proportion of ($\hat{n}_0$) seeds immigrating from the neighbourhood δ_i , and ζ is the proportion of (n_0) seeds of the i th cell emigrating to the neighbourhood δ_i . In MOSES ι and ζ are set equal. Similarly, Eq. 8 can be written as:

$$n(t+1) = M \times (n(t) + i(t) - z(t)) \quad (9)$$

with $i(t)$ and $z(t)$ indicating respectively the immigration and emigration vectors. Fig.VI.2 illustrates the life cycle model implemented in each cell of the CA.

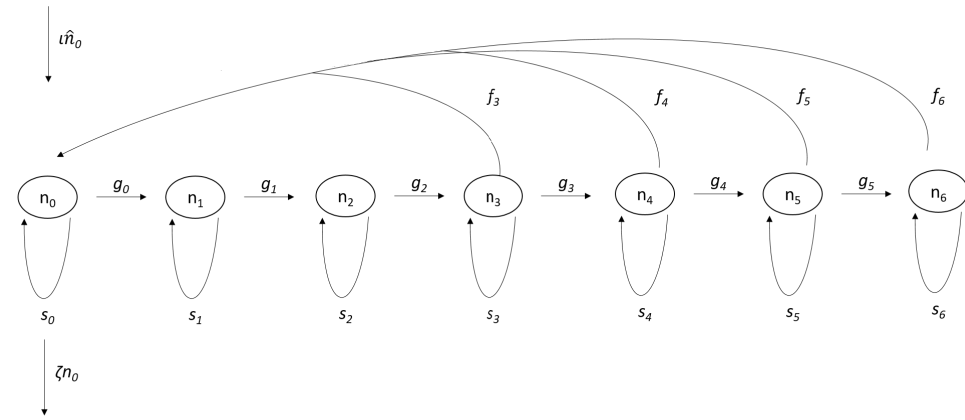


Figure VI.2: The tree life cycle implemented in each cell of the CA, with a size-structured life history. ι = proportion of immigrating seeds; ζ = proportion of emigrating seeds; n = tree population vector; s = probability to survive and not to grow to the next class; g = probability to survive and grow to the next class; f = tree fecundity rate.

The tree population in MOSES was constructed as being density dependent. There is in fact abundant evidence that intraspecific competition affects growth and survival of trees (Bontemps et al., 2013; He and Duncan, 2000). In MOSES we regulate the population growth rate using as maximum cell carrying capacity the total tree Basal Area (BA) per i th cell (Eq. 10; VanderSchaaf and Burkhardt, 2007):

$$BA_i = 7.85 \cdot 10^{-5} \sum_{j=0}^6 n_{ji} \cdot (d)^2 \quad (10)$$

where n is the number of tree in each size class j for each cell i , and d is the average diameter of the respective size class j , as given in Table VI.1. The cell carrying capacity

is then defined as a threshold of each cell total BA. If the total BA per cell exceeds such threshold then the population would shrink proportionally by a given constant⁵.

1/f noise and stochasticity in seed production

Modelling tree mortality in forest landscapes is a complex but necessary task, as tree growth models are very sensitive to the underlying mortality assumptions. However, tree mortality parameters are still poorly understood, due to the direct and complex dependence of tree survival to the individual genetic tree make-ups, environmental extremes, and competition with neighbouring trees (Monserud and Sterba, 1999). Often, only the latter is explicitly considered in growth models (Buchman et al., 1983). In order to improve the predictive ability of tree mortality models, Gertner (1989) outlines two possible approaches: (i) additional data collection, ideally using permanent plots, (ii) model restructuring to obtain mortality precision estimates. The former however critically depends on the representativeness of permanent plots, that should be able to observe both common and rare mortality events (e.g. fires, draught) frequently enough to build accurate predictions (Monserud and Sterba, 1999).

Modelling tree mortality requires, in theory, specifying within the model all environmental influences (meteorological, ecological, etc...) which are thought to impact mortality parameters. Nonetheless, deterministic models (where the model output is fully determined by all model parameters and its initial conditions) often fail to represent a proper ecological behaviour, which is usually perturbed by random events varying in space and time (Halley and Inchausti, 2004). Stochastic models embedding structured noises are thus being used in Ecology, as representations of all those environmental influences which cannot otherwise be explicitly included in the model. These methods proven to be effective in representing the complex nature of the environment, often reacting unpredictably to internal and external shocks. In ecological

⁵The average diameter for class-6 was set at 75 cm. The proportion of population loss when reaching the cell carrying capacity was set at 0.5.

studies, the most common noise model of environmental stochastic processes is the white noise⁶, a simple enough model of noise, which application has been criticised as a mere simplification of the complex ecological reality (Halley, 1996). Different "coloured" noises have been used in ecology to better represent how populations react to environmental fluctuations correlated in time (i.e. environmental shocks might have a different intensity in different time-scales) (Vasseur and Yodzis, 2004). The most important of these are $1/f$ noises, which are characterized accordingly to the shape of the noise spectral density. In $1/f$ noises, the variance changes with frequency according to an inverse power relationship: $1/f^\beta$ (for additional insights the reader can refer to Halley, 1996). Given this broad definition, white and also brown noises are special cases of $1/f^\beta$ noises (respectively with $\beta = 0$ and $\beta = 2$). This family of noise appears to well describe a wide range of environmental variables (Vasseur and Yodzis, 2004). Pink noise lies in between the two extremes, with a $\beta \approx 1$. This type of noise has been found suitable by former studies to represent ecological time series, as its variability continues increasing in time (Halley and Inchausti, 2004). An additional relevant property of the pink $1/f$ noise⁷, is its autocorrelation in time and space. This means that variability on timescales of 1-10 years are the same as that on timescale of 10-100 years, 100-1000 years and so on. Similarly for space, in a pink-noise process, the variability seen on a distance scale of 1km-10km is similar to what is seen on 100-1000km or any other decade of distance.

According to this line of reasoning, we found suitable to use $1/f$ noise to describe and model tree mortality as an environmental fluctuation. Following a $1/f$ noise approach, we can assume that environmental noise has a temporal and spatial correlation affecting (positively or negatively) tree mortality. In simpler terms this would entail that mortality of seeds could depend on common and rare events happening in different

⁶Noises are characterised by colours, which define the power spectrum of a given noise signal. In particular in the white noise, the variance is considered equally at all scales.

⁷From now on we will indicate with $1/f$ noise the pink noise only, as the $1/f^\beta$ noise with $\beta \approx 1$.

instant in time and on different time scales (temporal correlation), and that mortality in a given place is influenced by common and rare event happening in different places and spatial scales (spatial correlation). The use of $1/f$ noise to model ecological variability affecting tree mortality in different spatio-temporal scales is rather new. Moreover, it allows us to assess a relatively unexplored feature of the $1/f$ noise —spatio-temporal autocorrelation— in a single-species context.

In MOSES the parametrization of $1/f$ noise is embedded in seed mortality, which influences the population of the subsequent tree life stages. Given an environmental noise affecting seed mortality both in time and space ϵ_k , we denote with k the scale of the noise affecting the system both in time and space. In theory, k can assume values from 0 (the coarsest spatio-temporal scale under consideration) to ∞ (the smallest spatio-temporal unit). As follows, within the 2-dimensional lattice E in which MOSES is set, the area A affected by ϵ_k can theoretically be as large as E (A_0), and as small as one cell of E . Given that E is a squared grid, then A can assume only a finite numbers of values, corresponding to the factor r which partitions E into smaller squared sub-samples (Fig. VI.3):

$$A_k = \frac{1}{r^k} A_0 \quad k \in [0, 1, 2] \quad (11)$$

$1/r^k$ can thus be defined as the area interval impacted by the noise ϵ_k .

As described by Eq.8 above, the equation of population dynamics has a total of $(\omega + 1) \times (\omega + 1)$ parameters, each of which is subject to environmental variation. As we have argued, the most important parameter, which will dominate the fluctuation of the population is the seed mortality (Halley et al., 2018). In this study, we will concentrate on this parameter. In the case of trees, because of the huge fecundity values, there must be correspondingly large seed mortality values, as shown by Halley et al. (2018), leading to a highly asymmetric distribution of values, which leads is best described in terms of the mortality coefficient μ , which is related to the survival coefficient:

$$s_0 = e^\mu \quad (12)$$

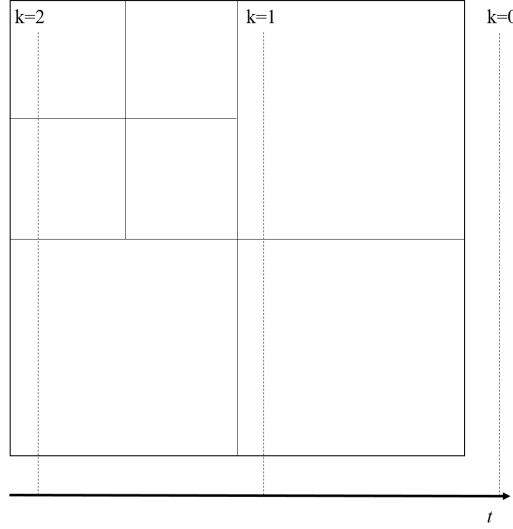


Figure VI.3: A visualization of the spatio-temporal scales used in MOSES. The figure shows how each k scale of the noise is associated to a different spatial unit (the size of the quadrants), and a different temporal unit t .

In situations of enormous pre-reproductive mortality, the survival coefficient will have an extremely asymmetric distribution, even when the distribution of mortality coefficient is normal or some other symmetric distribution. This coefficient consists of an average value plus a stochastic component, so that:

$$s_0(t) = e^{-\mu + \epsilon(t,x)} = s_0 e^{\epsilon(t,x)} \quad (13)$$

The stochastic component, is a function, not only of time but also of space, x in Eq. 13, since changes in mortality are not spread uniformly throughout the landscape. We consider this process to be a $1/f$ noise process. Assuming a spatio-temporal correlation of the environmental noise, the time interval τ of the noise ϵ with scale k can be set equal to:

$$\tau_k = \sigma \left(\frac{1}{r^k} \right)^\gamma \quad (14)$$

where σ is a scalar and γ indicates the correlation between spatial and temporal scales. Different values of γ refer to different strength relationships between the spatial and temporal environmental noises. In particular assuming value $\frac{1}{2} \leq \gamma \leq 2$, $\gamma=1$ would imply a linear relationship between space and time, while $\gamma=\frac{1}{2}$ and $\gamma=2$ correspond to

respectively a slower or faster reaction of the noise on the spatial scale. In MOSES γ was set equal to 1 as a short review of existing environmental phenomena show a linear relationship between time and space (Appendix L). Assuming in Equation 14 $\sigma=625$, a partitioning factor $r=25$, and $\gamma=1$, for values of k from 0 to 2, the spatial (A) and temporal (τ) intervals presented in Table VI.2 stands.

Table VI.2: The temporal and spatial scales implemented in MOSES.

K	Area (number of cells)	Time (years)
0	25^3	625
1	25^2	25
2	25^1	1

These are the spatial and temporal intervals which characterize how the environmental noise impacts the seed mortality in MOSES. Figure VI.4 illustrates the variation of the noise on the different spatial scales, showing snapshots in the spatial domain of the three processes (the first three panels), with levels of k equal to 2, 1, and 0, respectively, for a single instant in time. The panel in the outer right is the sum of these.

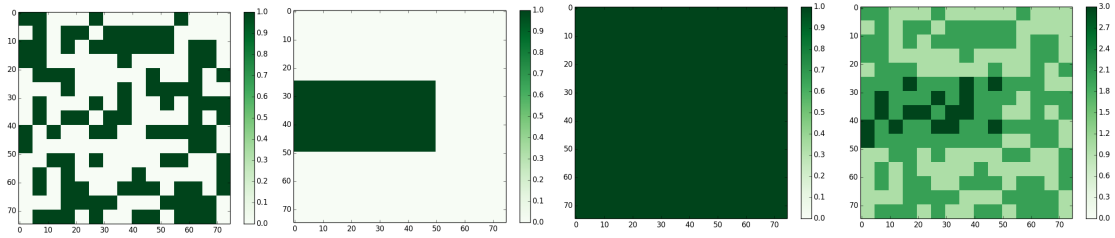


Figure VI.4: From left to right: spatial environmental noise for k equal to 2, 1, and 0 and respectively at time t . The last figure shows the sum of all noises within the CA spatial extent, which can assume values from 0 to 3.

Figure VI.5 shows the corresponding figures for the time domain, and the overall sum.

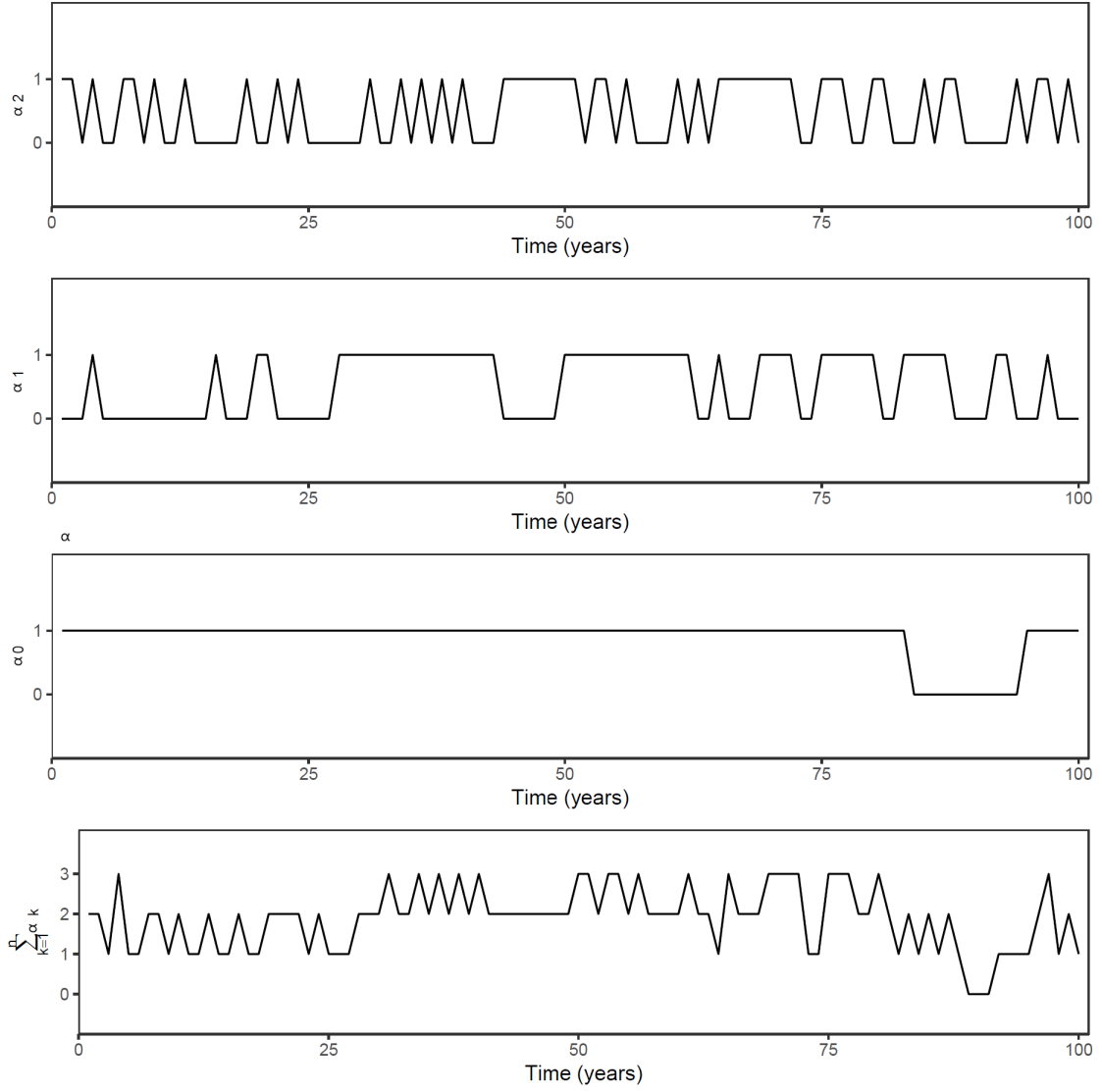


Figure VI.5: 100 years temporal variability of the environmental noise. The noise at the finest spatial scale (ϵ_2) has the fastest variation in time, while the noise at the largest spatial scale (ϵ_0) is the slowest in time. The last panel shows the overall sum of the three noise scales.

Given these temporal and spatial scales, the environmental fluctuation ϵ at each scale k for each i cell, is the result of an interaction between an α binary component, which can assume values of 0 and 1 only, and a ϕ component normally distributed with $\mu = 0$, and

variance⁸ = σ^2 :

$$\epsilon_{ki}(t) = \alpha_{ki}(t) \times \phi_{ki} \quad (15)$$

Practically speaking, α tells whether the shock occurs (i.e. $\alpha = 1$ indicates the occurrence of the shock, while $\alpha = 0$ indicates the opposite), while ϕ its magnitude. The probability of α to change state (from $0 \rightarrow 1$ or inversely from $1 \rightarrow 0$) will decrease as the time scale τ_k increases, as expressed in the following system of equations:

$$\begin{cases} \alpha_{ki}(t) = \alpha_{ki}(t+1) & \text{for } x < e^{\frac{-t}{\tau_k}} \\ \alpha_{ki}(t) = 1 - \alpha_{ki}(t+1) & \text{for } x \geq e^{\frac{-t}{\tau_k}} \end{cases} \quad (16)$$

Where x is a number drawn from a $[0,1]$ bounded uniform distribution. This means that the noise at smallest spatial scale ($k = 2$) will change more frequently as time passes, while the noise at the largest spatial scale ($k = 0$), will change less frequently as time passes (Fig. VI.5).

At time t , the seeds in each i cell will be impacted by the cumulative effects of noise ϵ_k for any value of k . If we indicate with S_0 the overall transition survival probability of seeds at time t (therefore $S_0 = s_0 + g_0$; see Eq.6), the adjusted overall seed survival rate $\tilde{S}_0$ impacted by the environmental noise can be express as follows:

$$S_0\tilde{(t)} = S_0 \cdot \exp\{\epsilon_0(t) + \epsilon_1(t) + \epsilon_2(t)\} \quad (17)$$

Under these specifications the effect on seed survival will depend by the cumulative magnitude that the overall noises will assume. In the simplest case in which $\epsilon_0 = \epsilon_1 = \epsilon_2 = 0$, the seed survival will be equal to its baseline. Finally $\tilde{S}_0$ is decoupled into the s_0 and g_0 components of the seed transition probabilities (Eq.4 and 5).

VI.2.3 The social layer

Social processes are prominent phenomena influencing the trajectories that an environment can assume in a socio-ecological system. Results from previous chapters have thoroughly illustrated how depopulation trends and management changes have

⁸The parametrization of ϕ 's variance is presented in Section VI.3.1.

impacted the vegetation layer in sacred forests and in their surrounding landscape matrix. In order to take into account the evolution of such social processes both in time and space, in MOSES a layer of social constraints has been imposed, defining the impacts three social phenomena have on the ecological layer of sacred forests: (i) human population flows, (ii) change in management practices, (iii) strength of prohibition rules in place.

We define Z a social layer with extent equal to the lattice E of the given CA as follows:

$$Z = f(P, W, F) \quad (18)$$

As eq. 18 shows, the social layer Z depends on the three social phenomena listed above. Specifically: (i) the village population, P , (ii) the main occupation of the labor force of the village (e.g. woodcutting, agriculture, sheep and goat farming, etc.), W , and (iii) the knowledge of the sacred forest and the strength of existing prohibition rules, F .

We represent the village population P through a time-varying vector that gives the village population in each year (t)⁹. The management status W tells whether each cell of the CA is actively managed or not by villagers. In MOSES this vector can assume only two values, which indicate the probability that the cell is managed (i.e. grazed by animals, with agricultural fields, etc.) and the complementary probability that the cell is not managed. These probabilities sum up to one. Similarly, F gives the probability that prohibition rules existing in the village about the presence of the sacred forests are active or not. As for the management status, this pair of probabilities sums up to one.

In MOSES we impose the following starting conditions. At time zero, the probability that prohibition rules F are active is equal to 1 (100% probability of active prohibition rules) in the forest surface, and zero otherwise. This means that at time zero all villagers are aware of the prohibition rules in place in the sacred forest. Moreover, we assume that both W

⁹The way in which this vector is build is presented in Appendix M.

and F depend on the village population P . Specifically:

1. P decreases at time $t + 1$ (depopulation scenario);

A decreasing human population indicates that less people are present in the village. This will in turn affect the management status W as there will be less people available to manage the land. This means that the probability of active management will decrease proportionally with the human population P .

2. P increases at time $t + 1$ (population-growth scenario);

In this case the reverse scenario will hold. An increase in human population will correspond to an increase in the probability of active management. Moreover, a decrease in human population affects the strength of the prohibition rules F , increasing the probability of no active rules to occur.

3. P remains stable at time $t + 1$ (steady scenario);

If the human population does not change, there is no effect on both management status and prohibition rules.

Finally, we assume that there is an interdependence between the management status W and the strength of the prohibition rules F : in cells where prohibition rules are existing, no management will occur. Figure VI.6 presents the decision tree which articulates how the social processes affect (or not) the vegetation structure of the model cells.

Whenever a certain management W occurs in a CA cell, then the tree population survival rate will be reduced accordingly. In particular, active management corresponds to three specific activities: (i) agriculture (if this is in place, survival probabilities for all tree size stages are equal to zero), (ii) woodcutting (if this is in place, larger tree size stages get their survival rates reduced, smaller size stages no), or (iii) pastoralism (which implies that smaller tree size stages get their survival rates reduced, larger size stages no). The way management types will change across years depend on secondary data and ethnographic research (Appendix M).

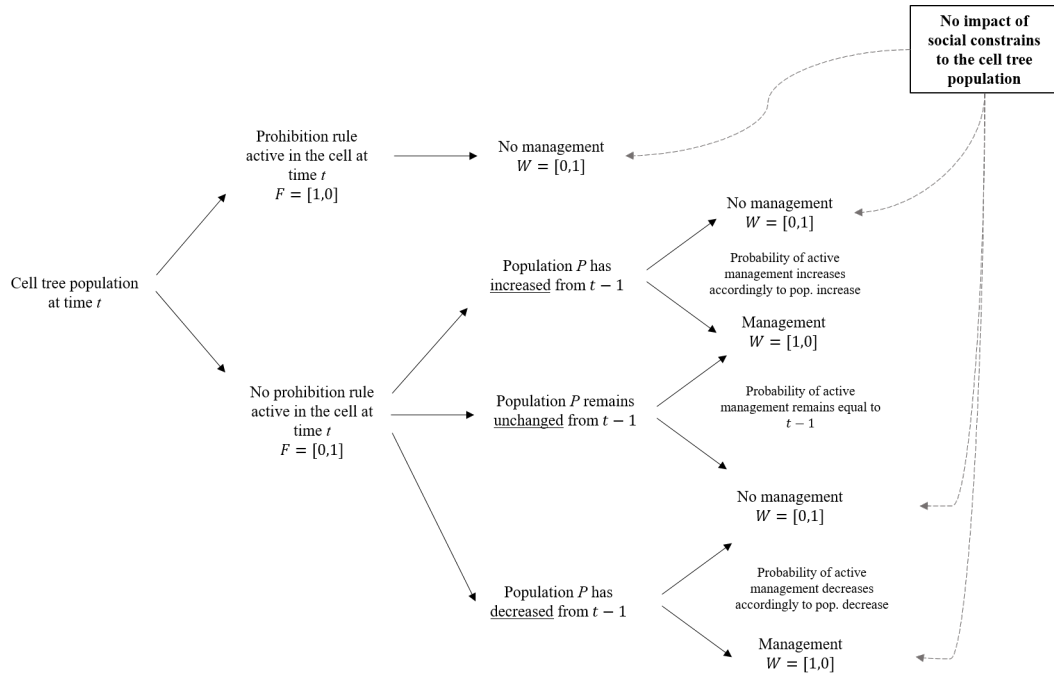


Figure VI.6: Decision tree showing how at each time t the social processes impact the vegetation of each model cell.

The effects of social processes is entered in the CA system via the tree mortality and survival transition probabilities of the Lefkovitch matrix of each i th cell (Eq.8). In particular, each cell i of the social layer S is characterized by the following [1-7] vector V :

$$V = (v_0, v_1, v_2, v_3, v_4, v_5, v_6) \quad (19)$$

This vector represents the overall effect of the social constraints on each stage of the tree life cycle (as Eq.3). Each year t , V impacts the survival probability of each cell i within the automaton as follows:

$$\begin{bmatrix} s_0 \cdot v_0 & f_1 & \cdots & f_{\omega-1} & f_{\omega} \\ g_0 \cdot v_0 & s_1 \cdot v_1 & \cdots & 0 & 0 \\ 0 & g_1 \cdot v_1 & \ddots & \vdots & 0 \\ \vdots & \vdots & \ddots & s_{\omega-1} \cdot v_{\omega-1} & \vdots \\ 0 & 0 & \cdots & g_{\omega-1} \cdot v_{\omega-1} & s_{\omega} \cdot v_{\omega} \end{bmatrix} \quad (20)$$

VI.3 Model parametrization

To verify the model functioning, MOSES is applied to one sacred forest: the excommunicated forest *Toufa*, belonging to the village of Greveniti (more information on the forest in Chapter II). This site is an ideal setting for testing the model for three main reasons. First of all, the sacred forest of Greveniti is mainly dominated by *F. sylvatica* individuals both at canopy and understorey level, allowing to test the CA model with single species population grow dynamics. Secondly, we have a large availability of social data on the village history and ecological data on the forest, which allowed to retrieved the initial socio-ecological parameters necessary for the model. Lastly, we have historical evidence of changing socio-economic processes across the years, which appear to have impacted the landscape surrounding the forest.

MOSES was parametrized with data stemming from three main set of sources: (i) sampled data, (ii) secondary data available in the literature, (iii) simulated data. Tree mortality and survival parameters for all stage classes (except n_0 , seeds) were derived from *F. sylvatica* stand curves of the 2014-2015 forest inventory (Chapter III). The normalized changes in slope between increasing diameter classes were assumed as tree mortality rates of each respective size class j . Tree growth parameters were obtained from available tree rings data sampled in Greveniti forest (Chapter IV). in particular the probability of a tree for a given size class j to grow to the next size class $j + 1$ was estimated as:

$$G_j = \frac{\overline{Gr}_j}{j} \quad (21)$$

where $\overline{Gr}$ indicates the average tree growth observed for rings within the size class j (Table 1). In other words G_j is directly proportional to the average tree growth, and inversely proportional to the size of the class. Seeds and seedlings were assumed to have a G probability of 1. Figure VI.7 graphically shows how G_j was computed. G_j and S_j probabilities were then used to compute g_j and s_j conditional probabilities (Eq.5). Fecundity rates were estimated from average seed production (Kg of seeds/ha) of

different *F. sylvatica* diameter size classes, available in literature (Lebret et al., 2001; Regina and Tarazona, 1999; Le Tacon and Oswald, 1977). Data from Drvodelić et al. (2011) was used to estimate the net weight of an average beech seed.

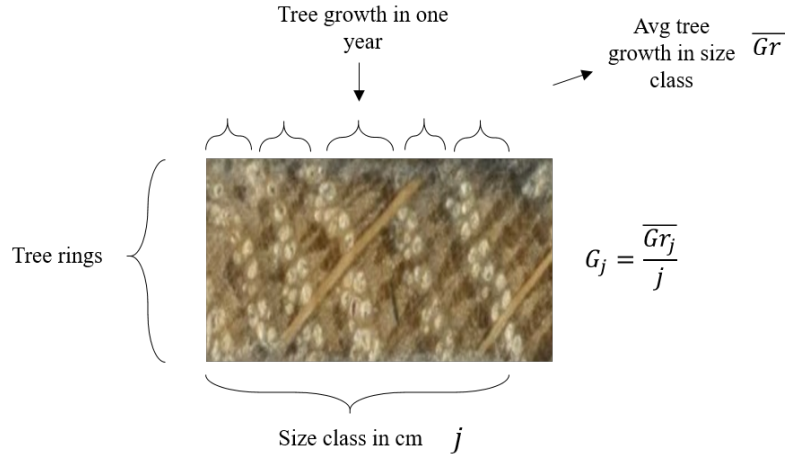


Figure VI.7: A graphical visualization showing how the tree growth parameter was retrieved from the tree rings data. Tree growth was measured for each ring within the corresponding size class. The average value was then calculated, which was divided by the corresponding size class in cm.

VI.3.1 Estimation of seed survival

Transition survival and mortality probabilities of seeds influence directly the population of the subsequent tree life stages. Their estimation is thus crucial for the validity of the model outcomes. However such probabilities are complex parameters to retrieve and unreliable estimations can have drastic impacts on population dynamics forecasts (Collet and Le Moguedec, 2007). Moreover in MOSES, seed survival is the parameter where environmental stochasticity enters the system. Therefore a proper estimation of its value and its relative variance is needed to assess the direct impact of the environmental noise on the tree population. A two-step estimation process was undertaken to estimate such rates. Firstly the equilibrium value of seed survival that makes the i cell Lefkovitch matrix converging to a stable stage population distribution was estimated. Then the variance of the seed survival rate was obtained using seedlings survival rates from the literature.

A generic population distribution model based on Lefkovitch or Leslie matrices, under reasonable circumstances, will converge to a single distribution (i.e. the Leslie's Convergence Theorem, Cull et al., 2000), if the transition probabilities of the matrix remain constant indefinitely (Caswell, 2001). The continued multiplication of the population vector by the matrix produces on the long-run a stable stage population distribution with constant proportion of individual per size classes, and finite growth rate. This latter estimate is defined as the asymptotic growth rate of the population λ_1 , independent from the initial distribution of the population. An estimation of λ_1 can be derived from the matrix *strictly dominant eigenvalue* (the eigenvalue which is larger in absolute value than any others). The interpretation of λ_1 can thus be used to observe whether the system population will asymptotically increase or decrease with time. In particular with $\lambda_1=1$ the population growth is equal to zero, with $\lambda_1 < 1$ the population growth is negative, while with $\lambda_1 > 1$ the population growth is positive. The rate r of population growth can be computed as follows:

$$r = \log(\lambda_1) \quad (22)$$

In order to obtain an estimation of the equilibrium rate of seed survival, the eigenvalues of a Lefkovitch Matrix of a generic i cell were computed for 50000 changing value of s_0 (values from 0 to 1 in a 50000 grid-interval), other things equal. For each iteration, the asymptotic growth rate λ_1 was stored. This process allowed identifying the seed survival rate yielding an asymptotic growth equal to zero ($\lambda_1=1$). Fig. VI.8 plots the results of the estimation process, identifying $s_0 = 9.5 \cdot 10^{-7}$.

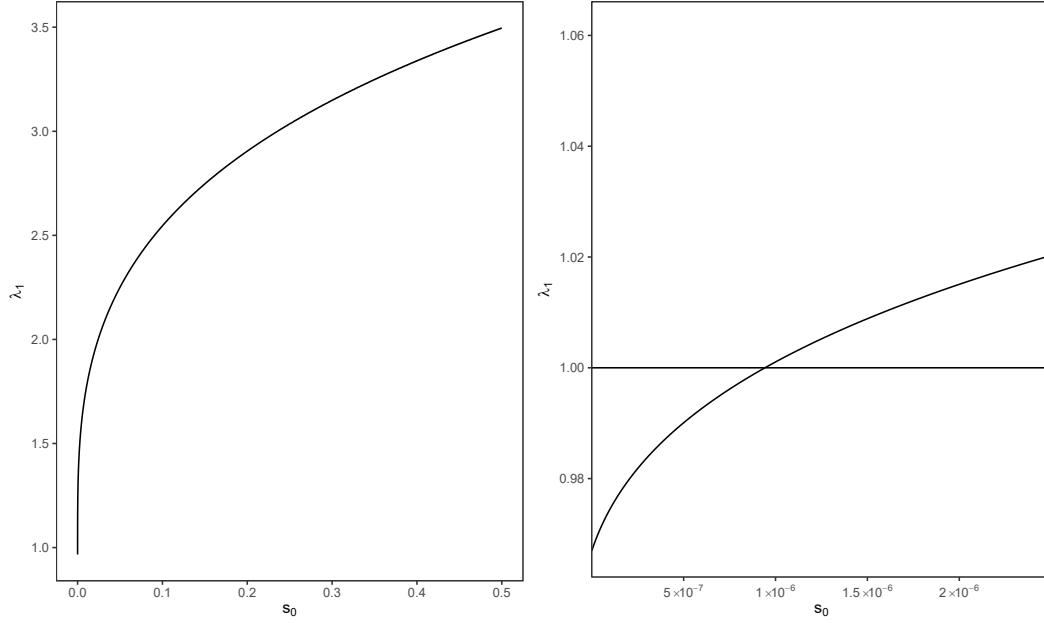


Figure VI.8: Changing value of seed survival S_0 and related effect on the strictly dominant eigenvalue of the transition matrix λ_1 . The horizontal line indicates the corresponding value of S_0 for $\lambda_1 \cong 1$. The right panel provides the same graph as presented in the left panel but with a smaller limit in the y-axis.

The variance of the noise ϕ component (Chapter VI.2.2) was estimated fitting a log-normal distribution to seedlings survival rates retrieved from the literature ($\sigma^2 = 3.199$; Bianchi et al., 2019; Szwagrzyk et al., 2001; Nakashizuka, 1988). Since the time-scale of the data considered spans approximately 30 years, we assumed the observed variance to be mainly representative of noises ϵ_2 and ϵ_1 varying respectively on a one or 25 years time-scales (Table VI.2). Their variances were thus assumed to be equal to half of the observed variance from the secondary data on beech seedlings ($\sigma^2 = 1.59$). The variance for the noise ϵ_3 was set at 0.3. Table VI.3 illustrates the main parameters used for the model fitting.

Table VI.3: Tree survival (s), growth (g), and fecundity (f) parameters used for the initial configuration of the tree population for the seven size classes (n_k) in each cell of MOSES. Parameters were estimated for *Fagus sylvatica*. Parameters s_0 and s_1 , indicating the probability of survival and no growth for seeds and seedlings were assumed as equal to zero following our assumption that all seeds and seedlings will either survive and grow or die.

Variable	n_0	n_1	n_2	n_3	n_4	n_5	n_6
s_j	0	0	0.49	0.65	0.89	0.92	0.97
g_j	$9.5 \cdot 10^{-7}$	0.1	0.16	0.23	0.06	0.02	-
f_j	0	0	0	$1.62 \cdot 10^4$	$6.55 \cdot 10^4$	$2.65 \cdot 10^5$	$8.73 \cdot 10^6$

VI.3.2 Initial configuration and hypothesis testing

The parametrization process presented in VI.3.1 allows us to estimate the following transition matrix for each cell of the CA model:

$$\begin{bmatrix} 0 & 0 & 0 & 1.62 \cdot 10^4 & 6.55 \cdot 10^4 & 2.65 \cdot 10^5 & 8.73 \cdot 10^6 \\ 9.5 \cdot 10^{-7} & 0 & 0 & 0 & 0 & 0 & 0 \\ 0 & 0.1 & 0.49 & 0 & 0 & 0 & 0 \\ 0 & 0 & 0.16 & 0.65 & 0 & 0 & 0 \\ 0 & 0 & 0 & 0.23 & 0.89 & 0 & 0 \\ 0 & 0 & 0 & 0 & 0.06 & 0.92 & 0 \\ 0 & 0 & 0 & 0 & 0 & 0.02 & 0.97 \end{bmatrix} \quad (23)$$

The initial tree population vector N^0 was constructed drawing random numbers from a Normal distribution truncated at zero with mean μ and standard deviation σ , computed after the truncation. These moments were computed using real data at plot level from the forest inventory described in Chapter II and III. Seeds at t_0 were randomly sampled from a bounded uniform distribution. The initial structured population of each cell was thus simulated by the following vector:

$$N^0 = [U [0, 300], N (38, 133), N (24, 47), N (8, 10), N (2, 4), N (1, 1), N (1, 1)] \quad (24)$$

N^0 was set as such only for the cell constituting the sacred forest. In outer zone, N^0 was set as a zero vector. The following figure illustrates the initial configuration of the CA.

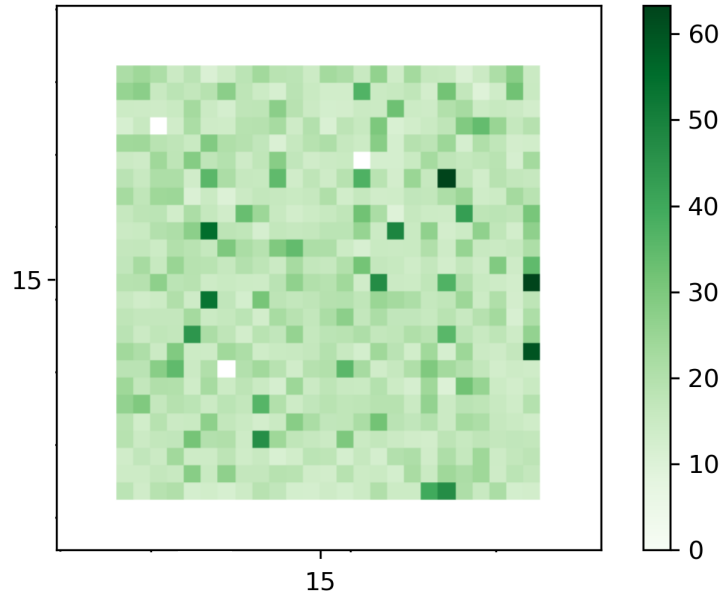


Figure VI.9: A visual representation of the initial configuration of MOSES. The green gradient represent the weighted average by size of trees in each model cell (in *cm*). Y and X axes represent the total extent in number of cells of the 25x25 initial forest.

The model defined above was used to quantify the effect that certain social processes acting upon sacred forests can have on the vegetation landscape matrix both in time and space. In particular three are the main scenarios which were assessed:

1. *Reduction in human population.* We assessed the rate of tree colonization in the landscape surrounding a sacred forest through time with decreasing human population, other things equal. The t_0 starting point of this simulation is an active sacred forest (with strong prohibition rules in its inner zones) embedded in a matrix of intensively managed landscape. We hypothesise that a steady reduction in population would be directly associated with an increase rate of tree colonization in the outer cells of the forest.
2. *Change in management practices (i.e. from agriculture to woodcutting).* We assessed the effect of a management change on the landscape matrix in combination with a

depopulation scenario, other things equal. Similarly to Scenario 1, the t_0 starting point is an active sacred forest (with strong prohibition rules in its inner zones) embedded in a matrix of intensively managed (=agricultural) landscape. In this scenario we hypothesise that population decline and management changes can have a compound effect on the increasing rate of tree colonization of the forest.

3. *Combined reduction in human population, management changes, and loss of knowledge.*

In this scenario, we assess both the rate of tree colonization on the outer zone of a sacred forest, and the inner structure of the forest, with deteriorating prohibition rules. The initial configuration is as above. The final hypothesis is that the three social processes interact one with each others in defining the spatial structure of the sacred forest in time.

To run the simulations, social data on population (P), management trends (W), and strength of prohibition rules (F) were retrieved from ethnographic research performed on site during 2019 and additional secondary sources (Efthimiou, 2019) (Appendix M). These three sets of data were used to parametrise the social layer of MOSES as presented in Chapter VI.2.3.

VI.4 Model checking

Two main validation strategies were run on the model outcomes. Firstly, we checked whether the model was able to generate stable population outcomes on the long-term, with varying social and environmental constraints. Secondly we performed a sensitivity analysis to verify how susceptible is the transition matrix asymptotic growth (Λ_1) to changing parameters.

Validation #1 Stability of model outcomes

The first validation aimed at checking the long term performance of the model. In particular, we wanted to assess how the model would behave, both in the absence of social pressures (thus a merely ecological model of tree population dynamics), and in

the absence of an environmental noise (a deterministic approach). Firstly, we would expect the model population to remain stable in the long term, proving that the parameters used to fit the trees population generate stable population outcomes. Secondly, we would expect a different pattern between the model subjected to noise and the one without, especially regarding the long-term variance of the tree population. Lastly, we would expect that the model subjected to only ecological processes would have different population outcomes compared to the one used including both social and ecological processes.

To this end three Monte Carlo experiments of 1000 iterations each were run. Each of them assessed the rate of tree colonization a simulated forest would reach in 300 years (i) with a stochastic ecological population model including a structured noise, (ii), with a deterministic population model in the absence of noise, (iii) with a stochastic ecological population model subjected to additional social constraints. The initial configuration of the forest was set as presented in Section VI.3.2. The results of this validation are displayed in Fig. VI.10.

This validation shows that the model generates stable outcome in the long term. In particular, on average, the forest increases by 0.02 ± 0.001 ha per year in its outer landscape in a 300 years time-span. Concerning the comparison with or without noise, the figure shows how the associated variability of the tree population dramatically changes in both specifications (first and second panels from the top). Under a stochastic regime with a $1/f$ noise, as expected, the variability increases as time passes, as the figure depicts. On the contrary, if we remove the environmental noise the variability is much more stable in time. Lastly, by comparing the model outcomes with or without the influence of social processes (first and third panels from the top), we observe that the variability is strongly reduced when we constrain the tree population by the observed historical trends of management, human population, and observance of existing prohibition rules.

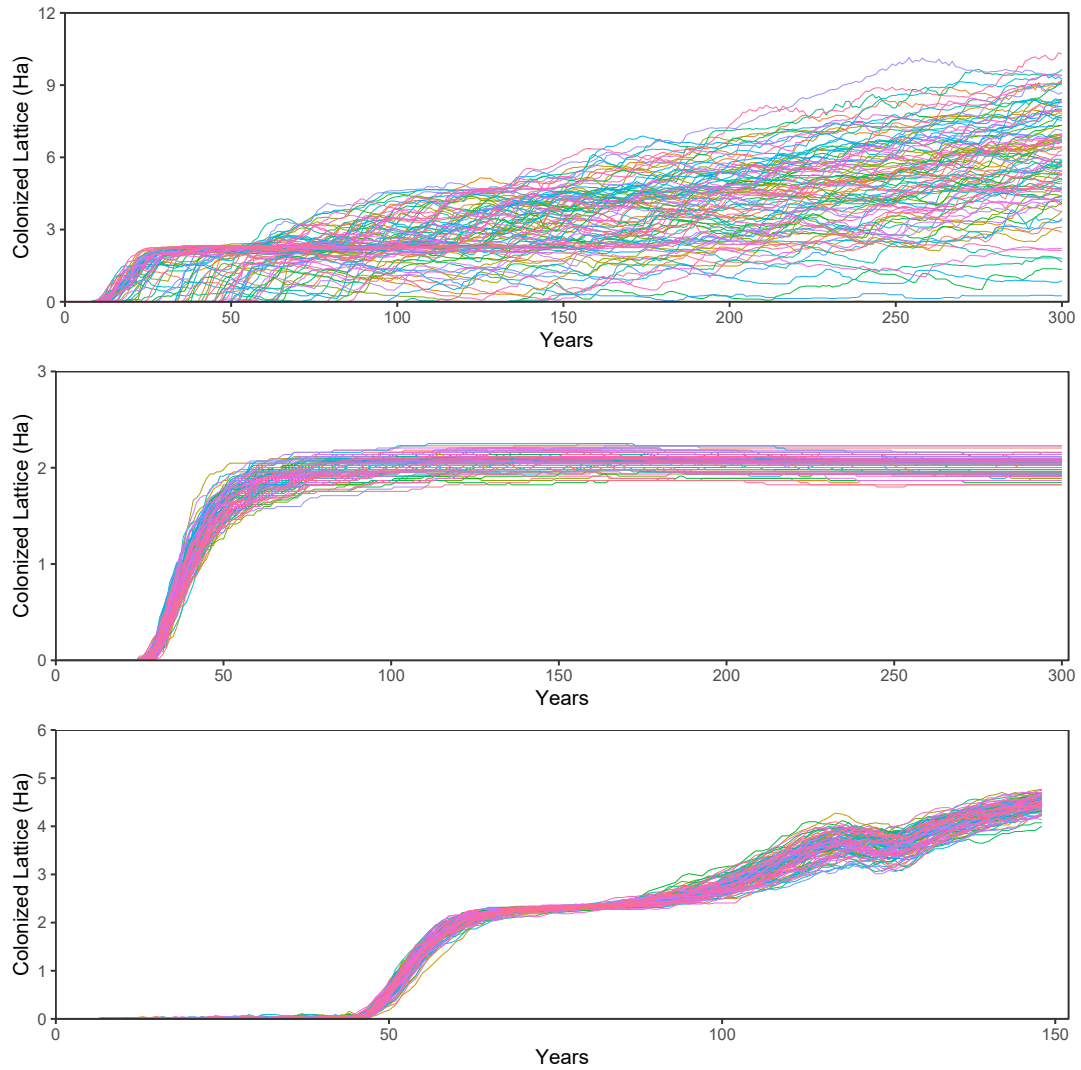


Figure VI.10: Tree colonization rate simulated across time with (from the topmost panel): (i) a stochastic ecological population model including a structured noise, (ii), a deterministic population model in the absence of noise, (iii) a stochastic ecological population model subjected to additional social constraints. In the last specification the time-series had a 148 years length only, due to the limited availability of social data . Each line represent a simulation. Only 100 simulations were plotted as to allow readability. Y axis shows the amount in ha of new cells colonized as forest over the years.

Validation #2 Sensitivity analysis of transitional probabilities

The second validation includes a sensitivity analysis to verify how susceptible is the transition matrix asymptotic growth (Λ_1) to changing parameters. We firstly computed analytical sensitivities and elasticities for the survival, growth and fecundity rates controlling the stage structure population model (Table VI.3). Analytical sensitivity

analysis reveals how a small changes in one of the Matrix parameters affect Λ_1 , other things equal. Analytical elasticity analysis transforms the sensitivity in a dimensionless rate, thus allowing comparative analysis among all transition probabilities (Donovan and Weldon, 2002). Elasticity values were finally recalculated for changing rate of the transition parameters, as to explore the overall sensitivity of the model to the established parameters.

The following table summarizes the analytical sensitivity and elasticity rates computed for all non-zero transitional probabilities listed in Table VI.3. The procedure used for calculating such rates follows Donovan and Weldon (2002).

Table VI.4: Sensitivity and elasticity rates of the *Fagus sylvatica* population in MOSES. Parameters s_0 and s_1 were not considered for this analysis as they have been assumed as equal to zero (all seeds and seedlings will either survive and grow or die)

Parameter	Sensitivity rate	Elasticity rate ¹
s_2	0.026	0.013
s_3	0.038	0.025
s_4	0.131	0.118
s_5	0.177	0.164
s_6	0.602	0.589
g_0	30108.8	0.013
g_1	0.129	0.013
g_2	0.079	0.013
g_3	0.056	0.013
g_4	0.206	0.013
g_5	0.202	0.013
f_3	$5.46 \cdot 10^{-10}$	$5.33 \cdot 10^{-6}$
f_4	$1.28 \cdot 10^{-9}$	$3.80 \cdot 10^{-5}$
f_5	$1.10 \cdot 10^{-9}$	$1.93 \cdot 10^{-4}$
f_6	$3.28 \cdot 10^{-9}$	$1.29 \cdot 10^{-2}$

¹The sums of all elasticity rates is equal to one, as they represent proportional sensitivity values.

These results indicate that, amongst all parameters, the survival and no growth rate s_6 has the largest elasticity value, thus having the strongest effect on the overall population growth (λ_1). This means that a 1% increase in s_6 will cause a 0.59% increase in λ_1 . As a

final check, all elasticity rates were recalculated for ten changing rates (from 0.1 to 1) of each transition parameter, other things equal. The results of this last sensitivity analysis are presented in Fig. VI.11.

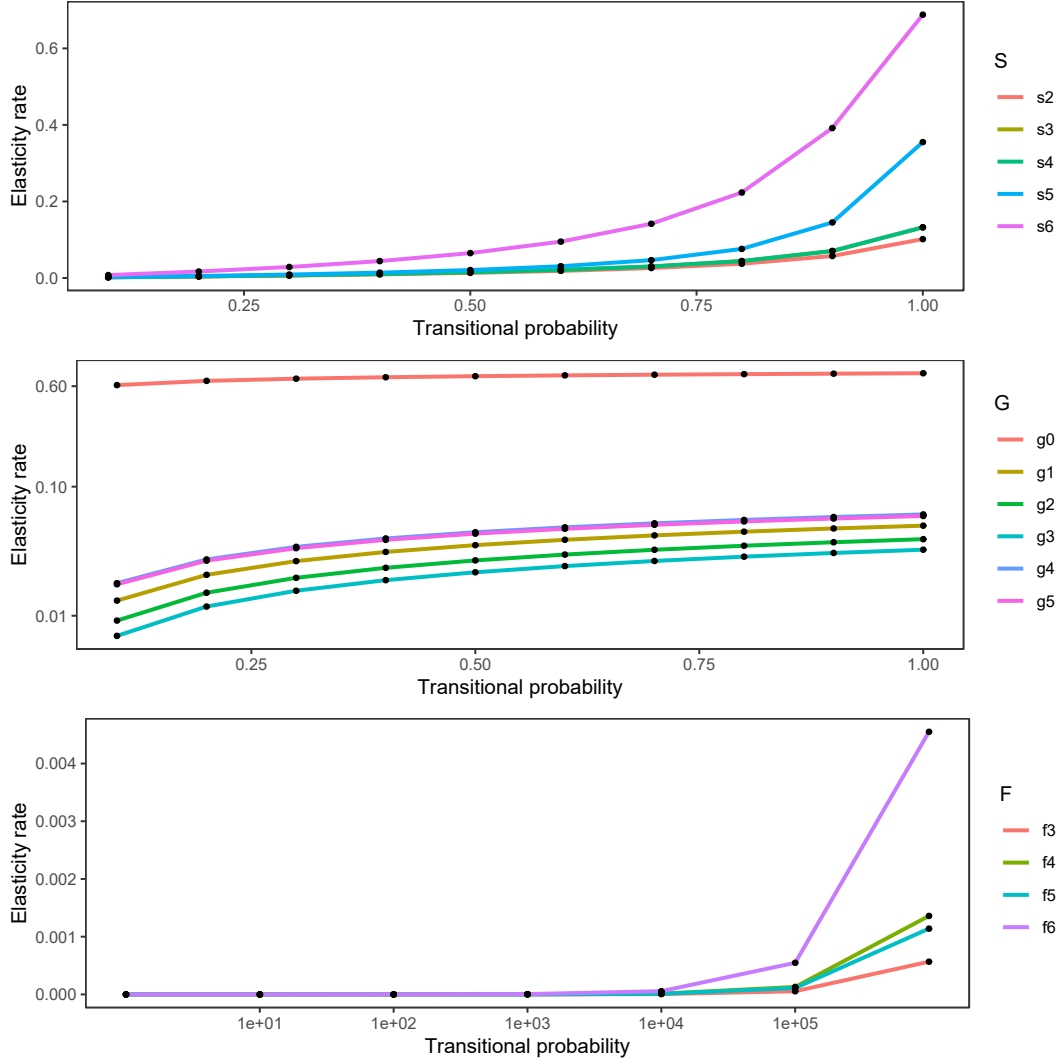


Figure VI.11: Elasticity values for varying transitional rates. Starting from the top: probability of survival and no growth S , probability of survival and growth G , fecundity rates F .

These results show that, as expected, the parameter yielding the highest elasticity value across different rates is the growth and survival rate for seeds g_0 , reason why extremely care has been taken in order to estimate its value. s_6 , on the other hand, is the parameter whose elasticity rate is the most sensitive to any change in the original transitional

probability. This effect is link to the fact that the largest trees are the ones who produce the highest amount of seeds, thus increasing overall the tree population growth.

VI.5 Results - Scenario simulations

Results of Scenario 1 are displayed in Fig. VI.12. The first graph is obtained from a Monte Carlo experiment (1000 iterations) used to simulate the impact of 148 years of human population decrease on the extension of the sacred forest surface, other things equal. The figure shows that a decrease in human population is correlated with an increase of the forest surface. On average, the sacred forest starts expanding during the first decade of the 20th century, after a drop in human population of about 25% (years 1874-1902). It then grows exponentially up until year 1960 (simultaneously to the largest population decrease observed, 45% drop between 1940 and 1951), when it becomes stationary for the next 35 years. In 1994 the forest start to expand again at a steady pattern. Overall the forest grew of 3.84 ± 0.01 ha (95% confidence interval), 27% of the forest area at t_0 . Figure VI.13 shows a spatial representation of one simulation at year 1868 ($t=0$), 1938 ($t=70$), and 2015 ($t=148$).

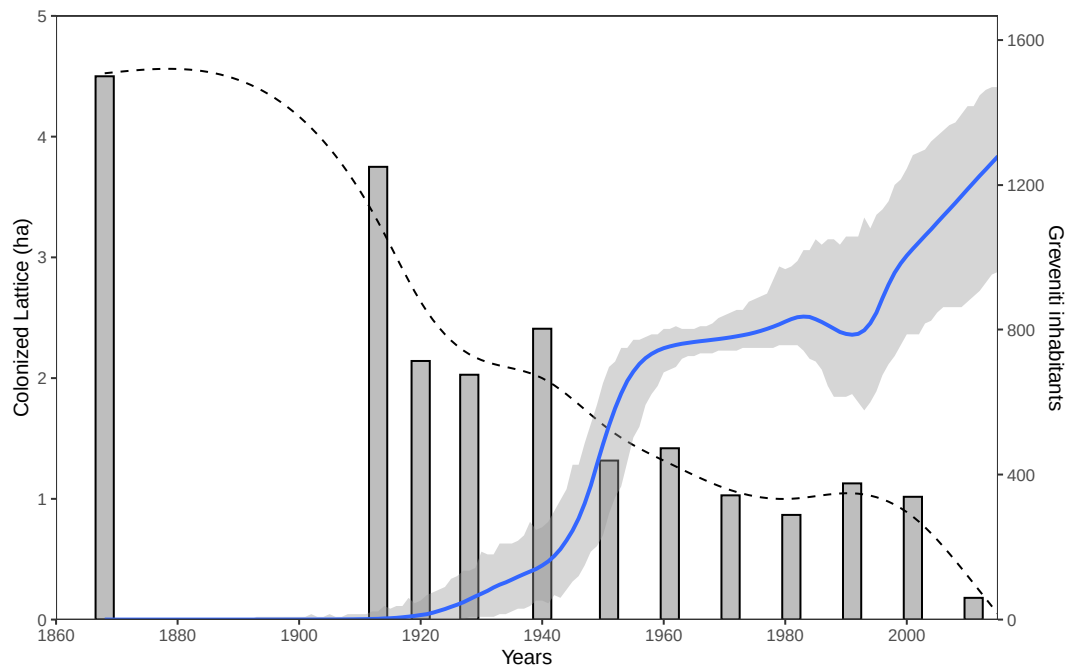


Figure VI.12: Monte Carlo experiment of 1000 iterations showing the effect of decreasing human population on the forest surface area. Each time series lasts 148 years (1868-2015). Y axis shows the amount in ha of new cells colonized as forest over the years. The black dotted line is the population data fitted to a GAM distribution (Appendix M, Fig.M1). The gray shading indicates the range between the simulated minimum and the maximum. On the secondary y-axis, population data at village level (bar-chart).

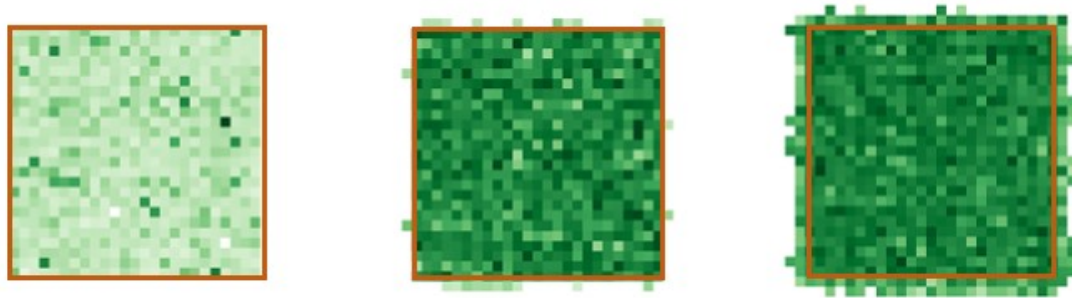


Figure VI.13: A spatial representation of the results of Scenario 1, showing the effect of decreasing human population on the extension of the forest surface. Panels represent the simulated forest (each cell representing the weighted average by size of trees larger than 5 cm) at $t=1$, $t=70$, and $t=147$ respectively. The squared grid represents the initial forest boundaries. In this simulation, the forest grows of 0.5 ha (4% of the forest area) in 70 years, and of 3.7 ha (26% of the forest area) in 148 years.

The combined effect of depopulation and change of management status (Scenario 2) is

shown in Fig.VI.14 and Fig.VI.15. Along with the population decline, the model considers the result of the multiple changes of management practices occurred in the time frame under investigation (i.e. from agro-pastoral activities to forestry, Appendix M, Fig.M2). The combined effect of population and management changes is as well correlated with an increase of the forest surface, with a steeper slope than the previous scenario. In particular the forest abruptly starts to increase exponentially in 1914, due to the combined effect of population decline and the loss of management in the area during the inter-wars period (years 1913-1949). Under Scenario 2, the forest grow of an additional 15% on average respect to Scenario 1, reaching 4.50 ± 0.01 ha (95% confidence interval) in total, 32% of the sacred forest area in t_0 .

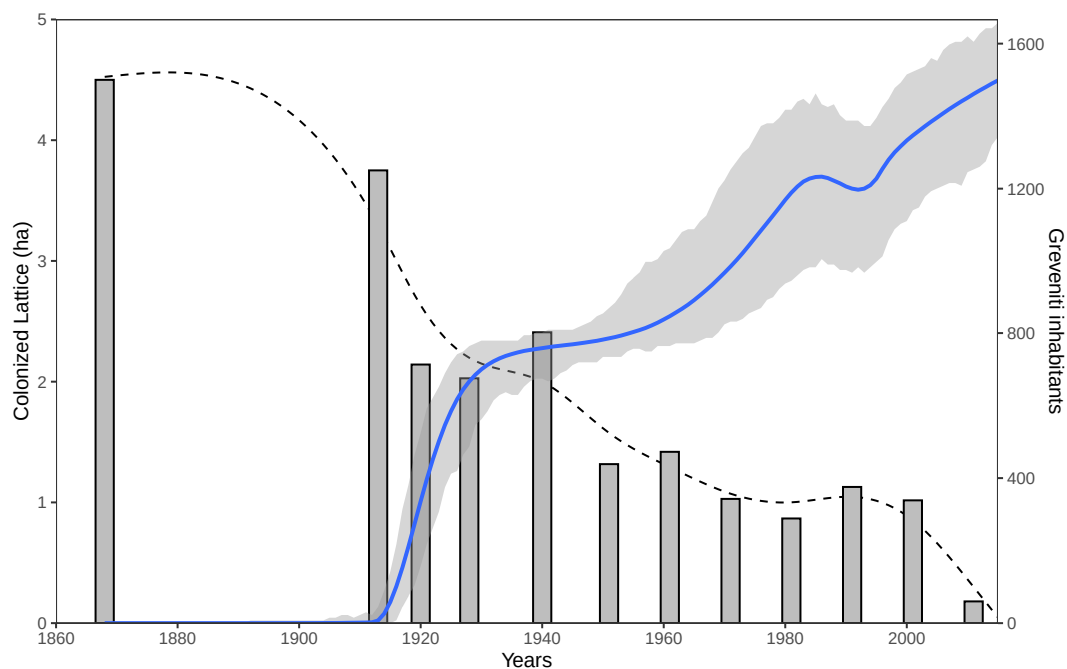


Figure VI.14: Monte Carlo experiment of 1000 iterations showing the effect of both decreasing human population and management changes on the forest surface area. Each time series lasts 148 years (1868-2015). Y axis shows the amount in ha of new cells colonized as forest over the years. The black dotted line is the population data fitted to a GAM distribution (Appendix M, Fig.M1). The gray shading indicates the range between the simulated minimum and the maximum. On the secondary y-axis, population data at village level (bar-chart).

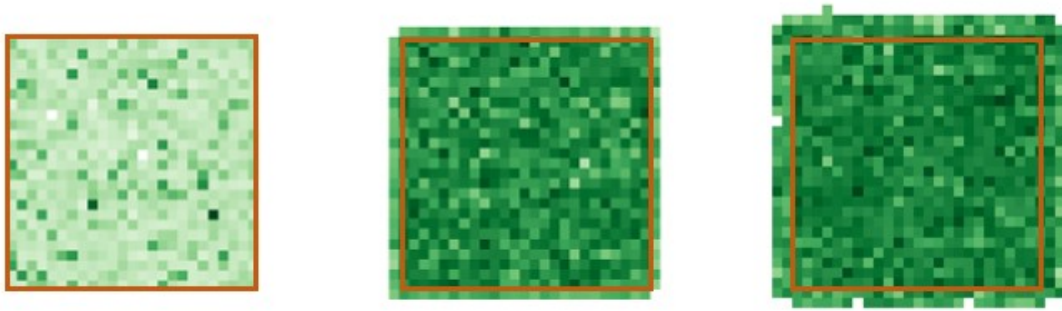


Figure VI.15: A spatial representation of the results of Scenario 2, showing the effect of both decreasing population and management changes on the extension of the forest surface. Panels represent the simulated forest (each cell representing the weighted average by size of trees larger than 5 cm) at $t=1$, $t=70$, and $t=147$ respectively. The squared grid represents the initial forest boundaries. In this simulation, the forest grows of 2.2 ha (16% of the forest area) in 70 years, and of 4.7 ha (33% of the forest area) in 148 years.

The last Scenario (3), is depicted in Fig.VI.16 and in Fig. VI.17. This time also the strength of the prohibition rules on the sacred forest (i.e. no cutting nor trespassing) is also allowed to vary in time. The graph shows that the combined effect of the three social processes maintains the forest increase on the outer landscape matrix equal to the former scenario, reaching in 2015 4.5 ± 0.01 ha. Additionally it shows that the foregone forest within the t_0 sacred forest boundaries peaks during the 1981-1995 interval (0.31 ± 0.01 ha). This is the combined effect of the deterioration of prohibition rules during mid 20th century and a close-to-stable population during the previous decade.

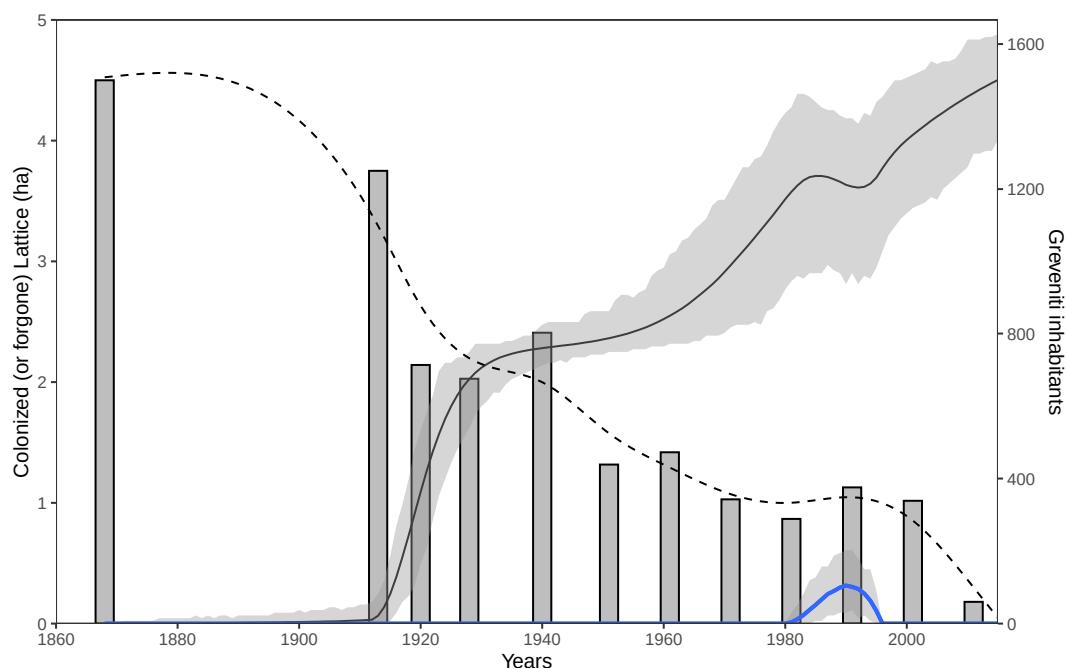


Figure VI.16: Monte Carlo experiment of 1000 iterations showing the effect of decreasing human population, management changes, and prohibition rules changes on the forest surface area. The blue line indicates the average yearly ha forgone within the sacred forest due to the loss of prohibition rules. The black line is the number of colonized ha on the landscape matrix due to population decrease and management changes. Each time series lasts 148 years (1868-2015). The black dotted line is the population data fitted to a GAM distribution (Appendix M, Fig.M1). Gray shadings indicate the range between the simulated minimum and the maximum of both distributions. On the secondary y-axis, population data at village level (bar-chart).

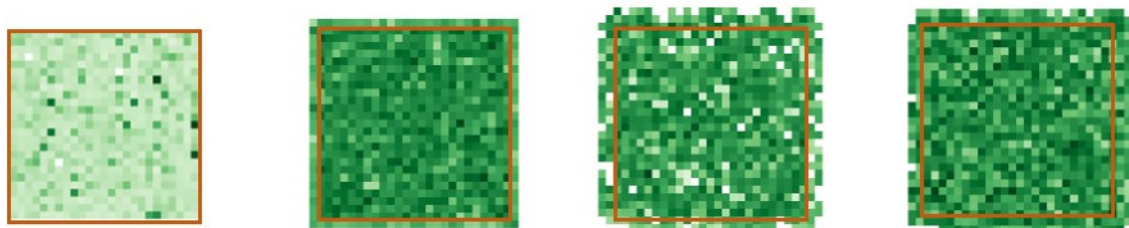


Figure VI.17: A spatial representation of the results of Scenario 3, showing the effect of decreasing human population, management changes, and prohibition rules changes on the extension of the forest surface. Panels represent the simulated forest (each cell representing the weighted average by size of trees larger than 5 cm) at $t=1$, $t=70$, $t=125$, and $t=147$ respectively. The squared grid represents the initial forest boundaries. In this simulation, the forest grows of 2.3 ha (16% of the forest area) in 70 years, and of 4.6 ha (33% of the forest area) in 148 years. Additionally the third figure shows how, due to deteriorating prohibition rules, part of the forest (0.27 ha) disappears.

VI.6 Discussion and conclusions

In this study we developed a CA model which was used to assess how sacred forests can react to changing social constraints both in time and space. In particular, we studied three main scenarios of changing population, management practices, and strength of prohibition rules over the extent of one sacred forest over a 150 years time span. Results of the scenarios are inspected and validated using existing secondary knowledge on the social processes which led to the formation of the sacred forests.

VI.6.1 Socio-economic changes and impacts on the sacred forests landscape

The simulations run show the key role of demographic decline in shaping the sacred forest landscape. In particular, our results indicate that depopulation only contributed to a 30% increase of the sacred forest surface in its surrounding managed landscape. This finding is in line with available secondary sources. Zomeni et al. (2008) analysed land-cover transitions obtained from satellite images in two areas of the Epirus region, observing an expansion of forest cover and shrublands in former croplands and grasslands ranging from about 1% to 16% during the 1945-1969 period, and from 2% to 24% during the 1969-1995 period. Overall croplands have diminished of 30% and 93% in the two study areas during the 1945-1995 period, in favour of forest areas. What also emerge is that, at landscape level, forest patches became less fragmented (Zomeni et al., 2008). This afforestation of the landscape strongly impacted sacred forests; once forest islands in a matrix of managed landscape, while today fused in a continuous forest areas of primary and secondary vegetation (Fig. VI.18). As additional studies show, the effect of depopulation and vegetation encroachment are deleterious also for several flora and fauna ecological communities (i.e. Zakkak et al., 2014).

Other studies in different areas of Greece also depict the strong linkage existing between historical population fluctuations and correspondent reduction in agricultural lands and number of active farms (Petanidou et al., 2008; MacDonald et al., 2000). These studies designate the II World War (1940-1944) and the subsequent Greek Civil War (1946-1949)

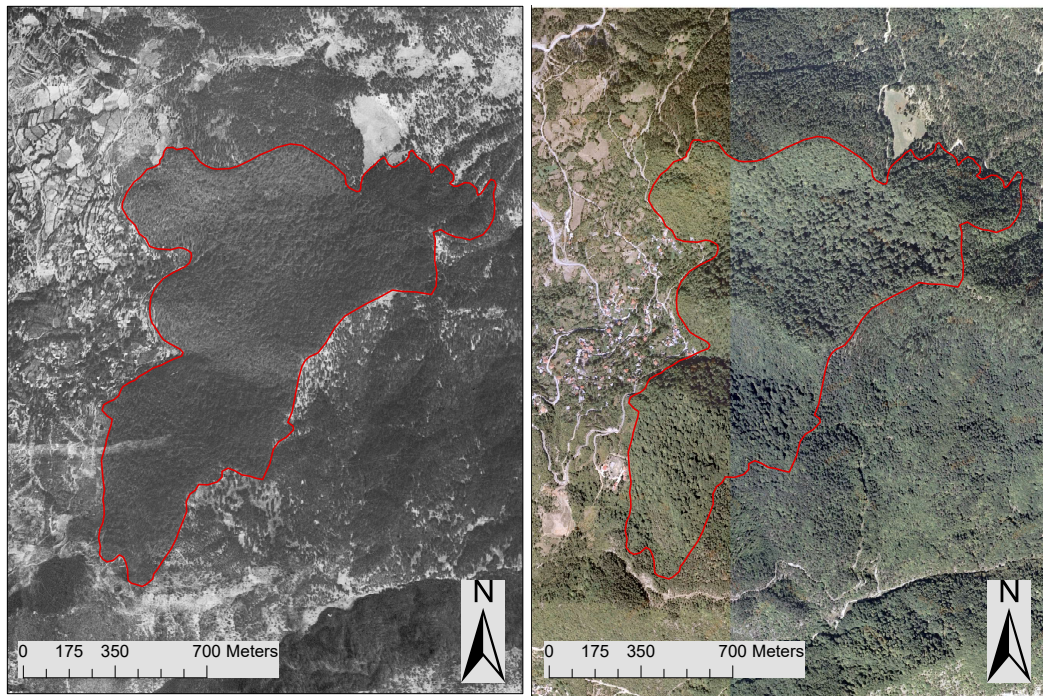


Figure VI.18: A 1945-2008 visual comparison of Greveniti sacred forest (red line) and its surrounding landscape.

as breaking points, during which the majority of the agricultural land was abandoned and left to natural succession patterns of vegetation. The effects of the wars on the landscape afforestation were well predicted by our model results (Fig.VI.12), where the highest increase in forest surface occurs just after the strongest population decrease (between 1940 and 1951).

Our model also shows that the combined effect of depopulation and management changes worsen the vegetation infilling in the outer areas of Greveniti sacred forest by an additional 15%. Traditional agriculture and animal husbandry ceased in Greveniti with the beginning of the wars period and the consequent abandonment of the village and its surroundings. Previous research shown that, up until the wars, agriculture with livestock farming were the main activities of the village (Efthimiou, 2019). After 1950, silviculture become the principal industry, thanks also to the growth of the secondary vegetation. Forestry is now the sector which currently gives jobs to the majority of the younger village inhabitants. The overall abandonment and management swift during

the inter-war period, increased the vegetation infilling potential in comparison to Scenario 1. Former scientific studies in the areas have highlighted that while depopulation can be considered the main driver of abandonment, changes in management and future management strategies are the factors that can have the widest effect on the socio-economic texture of the region. The identification of management priorities become thus key for the future conservation of these zones, i.e. tourism development alternatives as happening in many villages of the wide area of Zagori and Konitsa (Tzanopoulos et al., 2011).

Lastly, our model attempts to identify the effect of weakening prohibition regimes on the sacred forests. Previous research in the area have shown that in many villages the knowledge of the existence of sacred forests is fading; boundary elements are disappearing (see Chapter V), and the traditional relationship between society and the environment is changing (Stara et al., 2015b). If prohibition rules and cultural taboos are lost, then the socio-cultural construct which maintained partially intact the vegetation within sacred forests through centuries is also at risk. This can be even exacerbated if associated with a scenario of human population increase. The arrival in the area of tourists and neo-rural immigrants, unaware of the local traditions, might have an effect on the way the sacred forests are insofar managed and preserved. Our results indicate that a reduction in strength of prohibition rules affects the inner composition of the simulated sacred forest only in correspondence with close-to-stable (or growing) population; meaning that there exists more people, without knowledge of the sacred forests which might start trespassing it. On the basis of historical data, this situation might have occurred 30 to 40 years ago, during the 1981-1995 interval (Fig.VI.16). Ethnographic research has identified the occurrence of tree clearing activities within the Greveniti sacred forest dating back to about 50 years ago, coinciding with a lack of formal management rules over the sacred forest, which ceased with the renewal of the Forestry Ban on Post-War Protective Forest issued in 1972 (Efthimiou, 2019). This recorded trespassing activities partially match the simulated findings of the CA model.

VI.6.2 Model limitations

The CA interface developed successfully allowed to simulate the effect of changing historical scenarios on the sacred forest vegetation extent on the basis of available data. The use of a $1/f$ noise approach to simulate the effect of exogenous environmental shocks on the species population dynamics enabled to structure spatio-temporal environmental fluctuation in the system, partially bypassing the issue of lacking data on sensitive mortality parameters (i.e. seed survival). This approach proved to be successful in generating estimations which are aligned with literature data and secondary information, as presented in the former chapter.

Yet MOSES presents several limitations which should be closely tackled in order to allow its transferability to other socio-ecological systems. Firstly the model developed applies only to a single species tree population. This limits our ability to observe the combined effect of tree competition on the tree survival rates. Species competition in fact indicates how individual tree growth is influenced by intrinsic growth patterns of neighbouring species, which will have lastly an effect on the overall stand productivity (Venturoli et al., 2015; Le Bec et al., 2015), and the individual tree radial growth (Fernández-de Uña et al., 2015). Tree species competition is an essential process that occurs in forests, where limited environmental resources, such as nutrients, soil, sun and space, induces the individuals to compete, leading eventually some of them to be deprived (Begon, 1996). This is often the case when the system under investigation is a multi-aged or mixed-species forest. Forecasting competition is thus important as to understand plant-plant interactions and how such could affect the individual response to environmental variability. In models, competition is often integrated as a parameter of the tree growth curve. Additionally, management strategies might differ according to the species under investigation, indicating that an adjustment of the model to cover multiple species would be appropriate to extend its application to mixed-species sacred forests.

Secondly, sacred forests are constructs arising from the joined interplay of social, ecological and geophysical processes. The latter includes factors such as elevation, surface slope and aspect of the forest area, which are key for shaping and delimiting the spatial extent of the forests in the wider landscape. It is indeed true that in many cases sacred forests occur in areas with high slopes in the vicinities of villages, or in isolated hills in a general flat environment (Stara et al., 2016). The lack of a geophysical layer in MOSES, which can further affect the state of the system cells according to steepness or elevation patterns, is a limitation of this modelling approach. Its inclusion would generate more accurate predictions of the system extent, optimising the land use transition rules to observe complex spatial and topographic patterns of expansion in the landscape.

VI.6.3 Conclusion and management implications

The overall results of the simulations show that social and ecological processes acting upon sacred forests are in a delicate equilibrium which changes across years. 200 years ago, strong cultural taboos and management rules in villages with high demography contributed to generate stable socio-ecological constructs (the sacred forests). The evolution in time of certain social processes has modified this equilibrium, granting vegetation to expand in the outer zones of the forests where management has ceased, and threatening vegetation within the forest, when prohibition rules are fading.

The assessment of these scenarios constructed using a simulated model, allows practitioners to consider the impacts that different socio-economic processes have had on the cultural landscape, as to predict them in quantitatively form, always within the limiting assumptions assumed in the modelling interface. Spatially explicit models proved to be thus useful tools to verify existing oral narratives and local historical data of a cultural landscape, which are often partial and lack cross-validation (Norton et al., 2012). This is of great importance as it allows practitioners to verify the congruency of available narratives before developing *ad hoc* management strategies to protect and

conserve socio-ecological systems as sacred forests. Additionally, these models can assist policy makers in identifying the impact that future management scenarios might have on the cultural landscape, as for example the effects of a potential increase in population in rural areas and the subsequent impact this might have on the governance of sacred forests (Tzanopoulos et al., 2011).

Sacred forests narrate a strong tradition of communal management that has existed for hundreds of years, now fading due to socio-economic rural changes. There is yet no answer to which is the best management approach of sacred natural sites, due to the wide variety of ownership regimes and management systems currently existing in Greece and worldwide. In certain countries the inclusion of sacred groves within modern protected areas has led to severe environmental degradation caused by the abolition of traditional management systems (Zhou and Edward Grumbine, 2011), which are in many case the only factors responsible to their long-term resilience in time (Klepeis et al., 2016). Thus understanding whether, in view of rapid social changes, sacred forests will remain in the landscape and will continue to fulfil their central role for local communities is paramount for future conservation strategies.

The recent inclusion of sacred forests of Epirus into the UNESCO World Heritage List is a timely and valuable recognition of their religious, natural and cultural value. Recognitions however might bring up new challenges, as a possible increase of tourist flows in the area, which can contribute to a change in the way sacred forests are perceived and traditionally managed by the local communities (Barbhuiya et al., 2011). Models which can simulate the outcomes of complex management scenario, and the behavioural response of the community are thus important tools which can assist managers and communities as a whole towards a more comprehensive landscape management in socio-environmental systems as sacred forests.

CHAPTER VII

Concluding remarks

VII.1 Summary of the results

The overarching objective of this thesis was demonstrating that ecological and social processes interact reciprocally in forging SNS and sacred forests taking a selection of sacred forests in northwestern Greece as case studies. In particular, we focused on disentangling the relative influence of ecological and social processes in the composition, spatial patterning and temporal extent of SNS, as to improve current knowledge on the processes behind SNS formation and characterization in space and in time, a largely unexplored topic in current literature.

In the first application (Chapter III) we determined the relative effects of physical site environment and management regimes on the structure and composition of woody plant groups in six sacred forests in northwestern Greece. Our results indicate that species abundance and forest structure were found to be highly variable amongst the sites, with notable differences between canopy and understorey layers in certain sacred forests, which may be linked to past anthropogenic disturbances and expansion into abandoned farmland. Our main finding is that variation in species composition across sacred forests exist, and it can be linked, along with key environmental variables (such as site aspect), to the intensity and type of forest prohibition regimes (from areas with strict prohibition regimes entailing limited trespassing and wood extraction, to areas where grazing is allowed). In particular, historical prohibitions (primarily the control of grazing) are identified as one of the main determinants of the current structure of sacred forests.

Once identified the role of cultural practices in affecting the vegetation structure of sacred forests, in the second and third applications we further expand this analysis by looking to temporal and spatial dynamics existing across social and ecological processes in sacred forests (Chapter IV and Chapter V). In Chapter IV, we used dendrochronological techniques to date tree establishment events in five sacred forests in northwestern Greece, which allowed us to link distinct waves of recruitment of

canopy trees to common regional socio-economic changes. We estimate that the largest stand of canopy trees in the five forests are around 225-250 years old. A finding, which fits with available historical reconstructions. Additionally we identify two main common recruitment periods across the five studied sacred forests (second half of the 19th century and second half of the 20th century), which coincides with periods of human depopulation and events of anthropogenic turbulence and insecurity. This result indicates the strong-coupled interactions between human and ecological components of northwestern Greece sacred forests. In Chapter V, we looked at the other side of the coin, the effect of socio-ecological processes on the spatial extent of sacred forests. Focusing on sacred forests boundaries, we tried to understand how they are formed, taking into consideration also the role of peoples perception and taboos. Our results showed that we can assign features of the sacred landscape to the core of the sacred area (e.g. the location of the church, the location of the oldest tree communities), while other as peripheral to the sacred sites. Nonetheless, we also found that mapping the precise extent of a SNS is a complex endeavour due to the fuzzy and diffuse nature of their boundaries that depend on a multiplicity of factors, including peoples beliefs and perceptions, which dramatically change as time passes.

In the last application (Chapter VI), we bring all these findings together as to develop a model able to assess how the spatial extent of sacred forests can change due to varying socio-ecological constraints. This as to determining to what degree socio-economic processes (i.e. demographic and management changes) can affect the spatial extent of sacred forests otherwise determined by ecological processes of forest expansion. Ecological and social input data from the previous applications were used to develop a Cellular Automata model, which permitted us to simulate the expansion of a sacred forest over varying social processes in time (population flows, management changes, and strength of taboos and prohibition regimes). The results of the simulations show that two main processes —the reduction in population and the change of management practices— are fundamental social determinants on the way the environment react,

restricting or increasing the forest surface accordingly. The model we develop proved to be a useful tool to simulate the effect of changing historical scenarios on the sacred forest vegetation extent on the basis of available data, and we believe that its application can be extended to other cultural landscape and SNSs, as to verify oral narratives and local historical data.

After these analyses, we can conclude that it is possible to disentangle the relative influence of ecological and social processes in the composition, spatial patterning and temporal extent of SNS, provided that a set of ecological and social data on the sites are available. In the specific case of sacred forests in northwestern Greece, we have identified how the forests evolve dynamically in space as a reaction to a series of socio-ecological processes (expanding or shrinking in size), with a different magnitude over time. This as last can influence their internal tree composition and structure. Although ecological factors play a critical role in shaping the sites dynamics, we identified a series of social processes, which are key for defining the forests composition and spatial patterning, including prohibition rules and cultural praxis, management of the forest outer zone, and population trends. Unfolding these processes is fundamental for understanding how the historical change in cultural practices related to management of sacred sites has determined the trajectory of their vegetation composition and function.

VII.2 Sacred forests of Epirus: a dynamic socio-ecological system

In this thesis, we finally aim at conceptualizing SNSs using the Socio-Ecological Systems (SES) perspective. As early presented, SESs are complex human-nature systems where ecological, social, economic, and institutional realms interconnect both in time and space (Folke et al., 2005). These systems are characterized by persistent feedback loops between its components, and are additionally denoted by an intrinsic uncertainty of outcomes due to nonlinear behavioural responses of the systems actors. Linear approaches are

thus not suitable to assess reflexive responses of human and the environment to systemic changes. Schlüter et al. (2012), in particular, delineate a series of changing assumptions when apprising natural resource systems through the lens of SES (Table VII.1). In Table VII.1, we unfold the findings of this thesis as to identify the extent to which sacred forests, as SNSs, can be assimilated to the SES theoretical perspective. Our results indicate that SNSs should be conceptualized as SESs because they are:

- natural systems governed through community rules and cultural taboos (*linked systems of people and nature*), which have evolved as time passes and through subsequent ecological and societal changes (e.g. rural abandonment and consequent afforestation);
- systems determined by random processes, acting both on the social and environmental domains, which generate unique trajectories for each individual sites, nonetheless showing certain general macro patterns;
- systems in which multiple natural-social feedback loops exist which generate complex, nonlinear, and reflexive responses altering their architecture both in time and space.

Considering SNSs as SESs has multiple implications relevant from a policy perspective. First, unilateral policies that ignore the socio-ecological feedback loops of SNSs can have negative consequences on such systems (Anderies and Janssen, 2013). SNSs are not stable system but are subjected to multiple social and ecological changes. They thus require adaptive policy measures, which can comply with their socio-ecological changing context (Schultz et al., 2015). In the field of SNS, this can correspond to adaptive policy frameworks that depart from static sacredness to dynamic conservation strategies for the SNS and its socio-ecological values (a process that has been observed as informally happening in some SNS communities both in our study sites, and in SNSs in Myanmar, India, and Sri Lanka: Efthimiou, 2019; Weerasinghe, 2011). Secondly, policies on SNSs need to consider the interconnectedness between all elements systems,

Table VII.1: Different views and approaches when considering natural resources systems as SESs, and its adaptation to Sacred Natural Sites (SNS). The left column is taken from Schlüter et al. (2012). The right column is adapted incorporating the thesis findings.

Natural resource systems as SES	Sacred forests and SNS as SES
System dynamics exhibit thresholds, hysteresis	Present. SNSs current status depend directly on how it has changed in the past, both ecologically and socially.
Complexity and uncertainty of SESs are explicitly considered: probability distributions for key drivers and decision variables are highly uncertain, as are outcomes; some uncertainties are irreducible	Present. SNSs change in time and space due to both random and predictable processes.
Complex systems of interacting entities at microscale from which macroscale patterns emerge	Present. Each SNS is unique but general patterns can be elucidated.
Incorporate reflexive response of humans to forecasts and interventions	Present. Communities might react differently to adopted interventions (e.g. individuals accepting to trespass sacred sites).
Actors have imperfect knowledge, are boundedly rational or follow more complex decision patterns	Present. Communities linked to SNSs follow both individual beliefs as well as community-led rules and norms.
Management involves complex tradeoffs	Needed. Management of SNSs requires balancing out what is needed to do on the tangible landscape (e.g. forest management, ecosystems preservation) and what is needed to do concerning the social intangible constructs (people's beliefs).
Managed for resilience and adaptive capacity, management of stabilizing and amplifying feedbacks within a broader context	Needed. Management of SNSs implies adapting to changing natural and social conditions.

and between the micro-scale (each individual site, with its own specificities) and the macro, landscape, scale. From a conservation point of view, this means focusing on conservation networks of SNSs rather than on individual sites that can maintain and preserve the sum of their conservation values, in particular beta diversity (as proposed in Avtzis et al., 2018). This essential modularity of the policy framework dealing with SNS is necessary to allow the various actors and decision making centers (e.g. communities, local administrations) to be able to operate in a grid, but without hampering each other work (Anderies and Janssen, 2013). Lastly, policies dealing with SNSs should aim at being diversified and plurals. SESs rely upon multiple actors interactions. Maintaining alive the plurality of visions in SNSs is extremely important as it increases chances of social confrontations, possibly leading to the emergence of innovative ideas and practices linked to the management of SNSs.

VII.3 The way forward: future research and management considerations

Cultural ecosystem services have been identified as key nonmaterial values to support human wellbeing by many worldwide frameworks (e.g. the 2005 Millennium Assessment). Nonetheless, human-nature relationships are drastically changing. Rural-urban migration, abandonment of traditional lifestyles, and transformative processes of rural societies have impacted the way communities perceive their surrounding environment. If intangible cultural values disappear, then the socio-cultural force behind the SNS preservation is also at risk. This can weaken the effectiveness of landscape management practices that do not consider how cultural values in SNSs evolve through time. This is especially true when such values have spiritual and religious connotations linked to folklores, which are in disuse in modern societies, as in the case of sacred forests in northwestern Greece.

Understanding how spiritual values evolve and how they are perceived in modern societies become thus a valuable exercise. This as to explore future scenarios for the

management and conservation of spiritual and religious values attributed to sacred landscapes, investigating new uses for natural ecosystems that go beyond mere resources exploitation (e.g. aesthetics, recreational activities). Wild and McLeod (2008) identify six guiding principles for assisting managers and custodians in the conservation of SNSs: value recognition, integrated management, stakeholders participation, multidisciplinary, regulated access and use, and legal protection. In this sense, an innovative integrated management to conserve both values of Greek sacred forests (cultural and ecological) could be initiated, focusing on reinstating former management practices (e.g. low intensity grazing) to avoid further forest change. This would support retaining large-sized trees, one of the main iconic cultural symbols of the sacred forests, which are currently threatened by increased competition from tree regeneration. Revival of traditional management practices has been also proposed by Soliva et al. (2008) as a valid future scenario for Europe's mountain landscapes. Additionally, management of the boundary zone and the surrounding matrix could play a key conservation role in the near future, as forest fires, for example, could extirpate the rare forms of old growth wood pastures, especially those in the oak forest zone, which are now threatened with extinction.

Secondly, recognition of the role of local communities and their participation in the management of the sacred forests should also be promoted. The inclusion of a wide range of stakeholders into the management of the sacred forests and its boundary zones including the community custodians can enable the drafting of management plans legitimized by all relevant social groups, without interfering with existing cultural norms and taboos. This approach would allow the retention of the cultural and spiritual value of the forests in a changing environment and society, which would have a positive impact on key ecological conservation priorities (e.g. preservation of large-sized trees, and conservation of taxa that benefit from older trees). It will additionally empower the local communities providing them co-constructed management strategies for sacred forests, as well as enhancing regional development strategies for inner and

mountainous areas. The recent inclusion of Greek sacred forests in the UNESCO National Inventory of Intangible Cultural Heritage in 2014 (Intangible Cultural Heritage of Greece, 2015) creates opportunities and future challenges ahead for the management of this peculiar socio-ecological system.

Lastly, our findings suggest that each sacred forest, despite common ecological and social patterns, retains individual ecological features (e.g. species composition) and former ritual protections. This suggest that, in order to maintain and preserve the sum of the sacred forests' conservation values (in particular beta diversity), a large conservation network of these forest patches should be promoted, with the maintenance of distinct management regimes between the individual sites. As well as the direct benefits for landscape-scale biodiversity, this would increase socio-ecological resilience through promoting stronger local recognition and reducing the risk of complete loss of some components, e.g. due to fires (Avtzis et al., 2018) or, conversely, complete loss of disturbance by livestock.

VIII

Appendices

Appendix 1 (Chapter III)

A Canonical Correspondence Analysis

A Canonical Correspondence Analysis (CCA) was performed as a check of the robustness of the main NMDS ordination. The CCA was run on both plot and sub-plot data, in order to identify associations between the tree similarity matrices and a pool of environmental variables (Table I.1, Chapter I). In order to account for spatial autocorrelation, we performed a partial CCA controlling for the effect of spatial correlation by including in a third matrix the coordinates for each plot (the variable used to filter out the spatial correlation). This analysis was repeated for both plot and sub-plot data. The results of the CCA are displayed below (only the first two axes and significant environmental variables are shown).

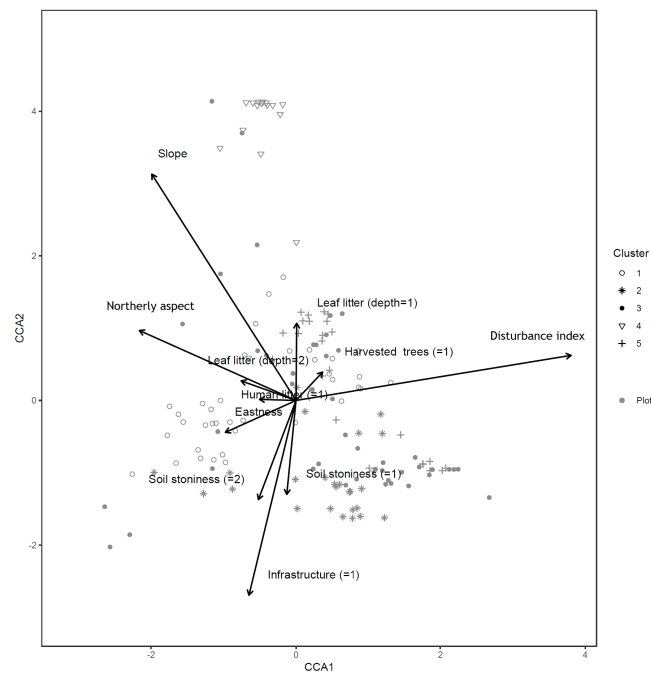


Figure A1: Canonical Correspondence Analysis of 152 plots across six sacred forests for canopy trees. Plots are labelled according to the five clusters produced by the cluster analysis (1= Pine (*Pinus nigra*), 2= Mixed oaks (dominated by *Quercus coccifera* and *Q. frainetto*), 3= Sclerophyllous species (*Acer monspessulanum*, *Q. coccifera* and *Juniperus oxycedrus*), 4=Beech (*Fagus sylvatica*), 5= Mixed broadleaves (*Q. pubescens* and *Carpinus orientalis*). Arrows indicate explanatory variables with their arrowheads indicating direction of increase.

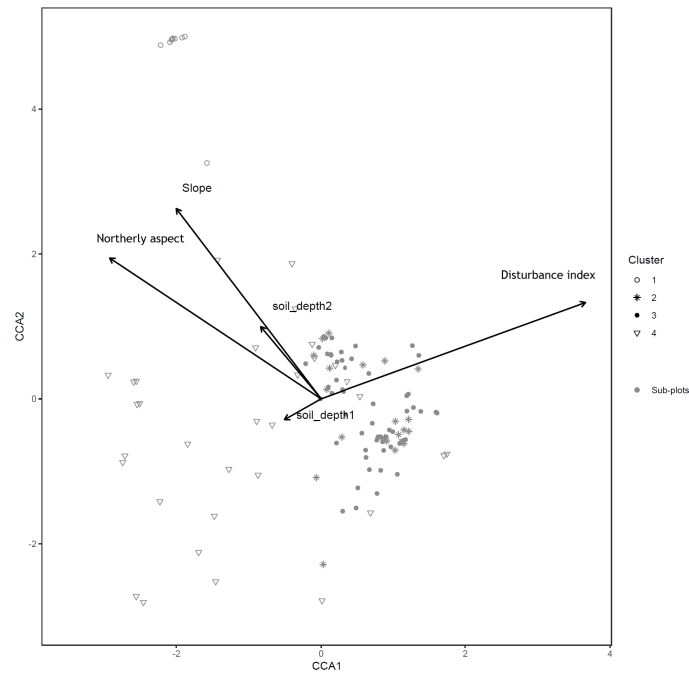


Figure A2: Canonical Correspondence Analysis of 140 sub-plots across six sacred forests for understorey saplings and shrubs. Sub-plots are labelled according to the four clusters produced by the cluster analysis (1=*Fagus sylvatica*, 2=*Carpinus orientalis*/*Acer monspessulanum*, 3=No significant species, 4=*Carpinus betulus*/*Ostrya carpinifolia*). The first axis of the NMDS ordination was transformed as $\exp(\text{NMDS})$ to improve readability. Arrows indicate explanatory variables with their arrowheads indicating direction of increase.

B Regression models between the three NMDS axis

Table B1: Regression models between the three NMDS axis scores and environmental variables for canopy trees (plot layer). Only variables significant at the $p < 0.05$ level are shown. The numbers in parenthesis represent the different levels of the ordinal categorical variables. Coefficients are unstandardized.

Explanatory environmental variables ^a	NMDS 1			NMDS 2			NMDS 3		
	Coeff.	<i>t</i> value	<i>p</i> value	Coeff.	<i>t</i> value	<i>p</i> value	Coeff.	<i>t</i> value	<i>p</i> value
Northerly aspect	-0.24	-3.79	<0.001	-0.16	-3.21	0.001	-0.13	-2.79	0.006
Slope angle	-0.01	-2.38	0.019				0.01	1.92	0.057
Disturbance index	0.28	2.23	0.028						
Infrastructure (=1)	0.18	2.10	0.038				-0.13	-1.90	0.059
Compact soil (=1)							-0.13	-1.94	0.054
Harvested or worked trees				-0.15	-1.86	0.050	-0.21	-2.67	0.009
R ²	0.74			0.63			0.50		

^a Explanatory variables are defined in Table 3 in Chapter 2.

Table B2: Regression models between the three NMDS axis scores for understorey saplings and shrubs (sub-plot layer) and plot environmental variables. Only variables significant at the $p < 0.05$ level are shown. The numbers in parenthesis represent the different levels of the ordinal categorical variables. Coefficients are unstandardized.

Explanatory environmental variables ^a	NMDS 1 ^b			NMDS 2			NMDS 3		
	estimate	z value	p value	Coeff.	t value	p value	Coeff.	t value	p value
Intercept				-0.09	-2.17	0.032			
Northerly aspect	-0.20	-3.40	0.001						
Slope angle	-0.32	-5.51	<0.001						
Altitude	-0.22	-3.88	0.000						
Leaf litter depth (=0)	0.22	3.27	0.001						
Leaf litter depth (=1)							-0.04	-2.03	0.045
Leaf litter depth (=2)							-0.07	-2.48	0.014
Vegetation ground cover							-0.06	-1.98	0.051
Deadwood abundance (=0)	0.32	4.72	<0.001						
Deadwood abundance (=2)	-0.14	-2.03	0.04						
Soil compaction	0.37	-5.49	<0.001						
Soil depth (=1)	0.29	4.35	<0.001	0.05	2.29	0.058			
Soil depth (=2)	-0.15	-2.23	0.026						
Soil stoniness (=2)							-0.08	-3.80	0.000
Easterly aspect							0.03	2.05	0.043
Infrastructure				0.04	2.28	0.024			
Disturbance index	0.44	7.11	<0.001				0.05	2.57	0.011
Human litter							-0.05	-2.30	0.023
R ²	n/a			0.32			0.61		

^a Explanatory variables are defined in Table 3 in Chapter 2.

^b Testing of the regression assumptions showed a non-linear distribution of model residuals, therefore the Kendall rank correlation coefficient was used (estimate, z test, and p value).

C NMDS ordination: all axes combinations

In this appendix all the remaining graphical representations of the NMDS analysis at plot and sub-plot are provided.

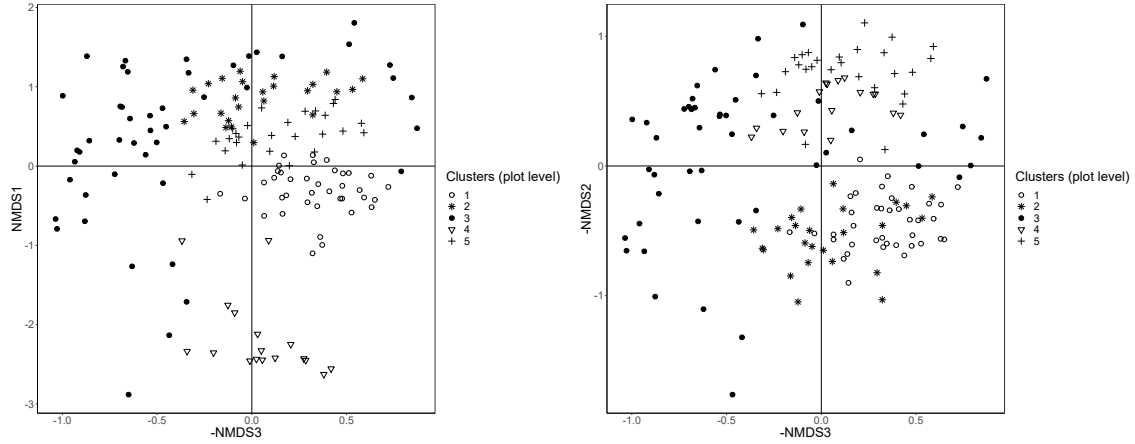


Figure C1: Plot data - NMDS ordination (axes 1 and 3 left panel; axes 2 and 3 right panel) of 154 plots across six sacred forests for canopy trees. Plots are labelled according to the five clusters produced by the cluster analysis (1= Pine (*Pinus nigra*), 2= Mixed oaks (dominated by *Quercus coccifera* and *Q. frainetto*), 3= Sclerophyllous species (*Acer monspessulanum*, *Q. coccifera* and *Juniperus oxycedrus*), 4=Beech (*Fagus sylvatica*), 5= Mixed broadleaves (*Q. pubescens* and *Carpinus orientalis*)).

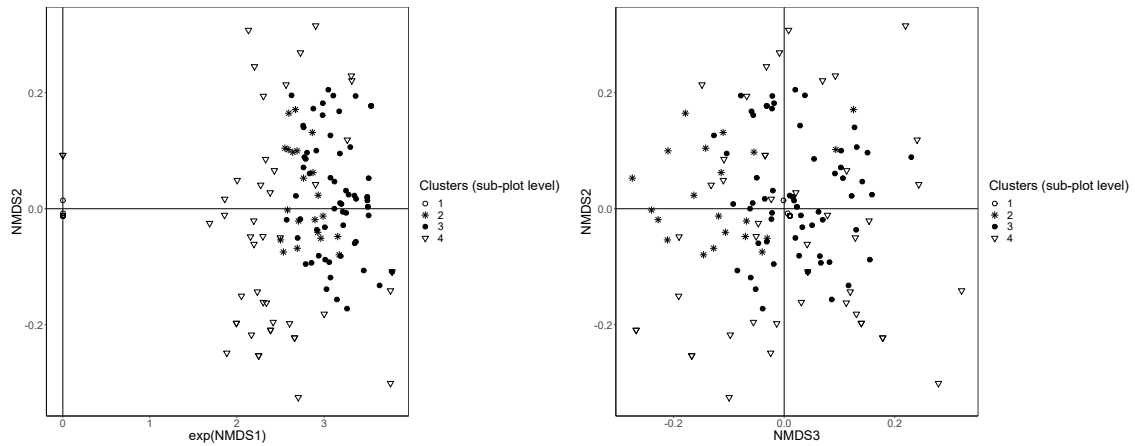


Figure C2: NMDS ordination (axes 1 and 2 left panel; axes 2 and 3 right panel) of 147 sub-plots across six sacred forests for understorey saplings and shrubs. Sub-plots are labelled according to the four clusters produced by the cluster analysis (1=*Fagus sylvatica*, 2=*Carpinus orientalis*/*Acer monspessulanum*, 3=No significant species, 4=*Carpinus betulus*/*Ostrya carpinifolia*). The first axis of the NMDS ordination was transformed as $\exp(\text{NMDS})$ to improve readability.

D Kruskal-Wallis non-parametric test, plot layer

Table D1: Kruskal-Wallis non-parametric test on selected environmental variables for the five clusters of canopy trees (plot layer) defined by the NMDS analysis of canopy trees in Figure 2, Chapter 3. Letters indicates groups that do not significantly differ within rows ($p < 0.05$). χ^2 = chi-square; *d.f.* = degrees of freedom (between-groups degrees of freedom and within-groups degrees of freedom).

Environmental variables	Clusters					Kruskal-Wallis non parametric test results		
	1	2	3	4	5	χ^2	<i>p value</i>	<i>d.f.</i>
Northerly aspect	a, b	a, b	a, b	a	b	12.5	0.014	4;149
Easterly aspect	a, b	a, b	a, b	a	b	19.9	0.001	4;149
Slope angle	a	b	b	a	b	45.1	<0.001	4;149
Altitude	a	b	b	a	b	52.3	<0.001	4;149
Leaf litter depth	a	a, b	b	a, b	a, b	20.7	0.000	4;149
Deadwood abundance	a, b	a, b	a	b	a, b	12.6	0.014	4;149
Soil compaction	a	b	b	b	a	22.7	<0.001	4;149
Soil depth	a, b	a	a	b	a, b	18.5	0.001	4;149
Disturbance index	a	b	b	a	b	41.6	<0.001	4;149

E Kruskal-Wallis non-parametric test, sub-plot layer

Table E1: Kruskal-Wallis non-parametric test on selected environmental variables for the four clusters of understorey saplings and shrubs (sub-plot layer) defined by the NMDS analysis of understorey in Figure 3, Chapter 3. Letters indicates groups that do not significantly differ within rows ($p < 0.05$). χ^2 = chi-square; $d.f.$ = degrees of freedom (between-groups degrees of freedom and within-groups degrees of freedom).

Environmental variables	Clusters				Kruskal-Wallis non parametric test results		
	1	2	3	4	χ^2	<i>P value</i>	<i>d.f.</i>
Northerly aspect	a	b	b, c	a, c	21.8	<0.001	3; 143
Easterly aspect	a	b	a, b	b	11.5	0.009	3; 143
Slope angle	a	b	b	a	27.8	<0.001	3; 143
Altitude	a	b	b	b	27.1	<0.001	3; 143
Leaf litter depth	a	a	b	a, b	15.8	0.001	3; 143
Deadwood abundance	a	a, b	b	a, b	15.2	0.002	3; 143
Soil compaction	a	a, b, c	b	c	28.6	<0.001	3; 143
Soil depth	a	a, b	b	a, b	15.4	0.001	3; 143
Disturbance index	a	b	b	a	39.7	<0.001	3; 143
Infrastructure	a	b	b	a	3.88	<0.001	3; 143

F Final reconfiguration of clusters

Table F1: Final reconfiguration of canopy tree (main plot) and tree seedling and shrub (sub-plot) clusters. The numbers inside the boxes refer to the number of sample units (plots and sub-plots) corresponding to each combination of clusters. The numbers of unclassified sample units and the total numbers are shown outside the boxes. Double lined boxes show the final configuration of four woody plant groups defined in Figure 5: A, HIGH-SAP; B, HIGH-SHR; C, LOW-SAP; D, LOW-SHR.

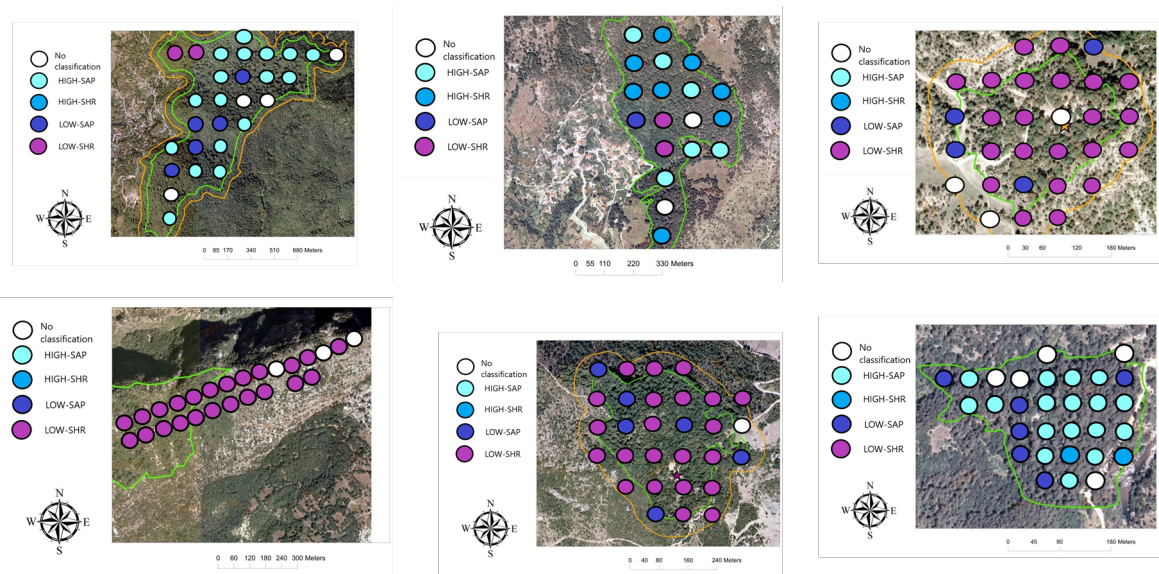
Cluster analysis - plot layer	Cluster analysis – sub-plot layer						Total sample units		
	Cluster 1		Cluster 4	Cluster 2		Cluster 3		Unclassified sub-plots ^a	
	Cluster 1	A	2	23	B	1		9	3
	Cluster 4		12	3		0		0	2
	Cluster 2	C	0	8	D	2		14	1
	Cluster 3		3	7		5		27	5
	Cluster 5		0	4		12		11	0
	Unclassified plots ^a		0	1		0		3	3
	Total sample units		17	46		21		63	14
	^a The 18 unclassified sample units include 4 plots with no trees, 11 plots with no understorey saplings or shrubs, and 3 with neither trees nor understorey.								

This second table shows the distribution across sites of the plots classified into the final four woody plant groups:

Table F2: Distribution across sites of the plots classified into the final four woody plant groups.

Village	HIGHSAP	HIGHSHR	LOWSAP	LOWSHR	Unclassified
Greveniti	17		5	2	4
Palioseli	6	8	1	2	2
Vovoussa	17	2	6		5
Vitsa			4	22	3
Kapesovo				23	3
Mazi			6	22	1

Lastly, the six maps below, shows how the woody plant groups are spatially located on the six sacred forests. (top row from the right: Greveniti, Palioseli, Vitsa; bottom row from the right: Kapesovo, Mazi, Vovoussa):



G Environmental variables across groups

Table G1: Average values of selected environmental and forest structure variables for the four woody plant groups. Letters indicate groups that do not significantly differ within rows ($p < 0.05$; Kruskal-Wallis non-parametric test). χ^2 = chi-square; *d.f.* = degrees of freedom (between-groups degrees of freedom and within-groups degrees of freedom).

Environmental variables	Woody plant groups				Kruskal-Wallis non-parametric test results		
	HIGH-SAP	HIGH-SHR	LOW-SAP	LOW-SHR	χ^2	<i>P</i> value	<i>d.f.</i>
Northerly aspect	0.12 ^a	-0.48 ^{ab}	-0.02 ^{ab}	-0.35 ^b	16.2	0.001	3; 137
Slope angle (°)	31.3 ^a	31.0 ^{ab}	22.7 ^{bc}	18.2 ^c	45.6	<0.001	3; 137
Altitude (m)	1157.2 ^a	1193.6 ^a	935.0 ^b	911.9 ^b	39.1	<0.001	3; 137
Leaf litter depth	1.20 ^a	1.20 ^{ab}	0.86 ^{ab}	0.78 ^b	13.8	0.003	3; 137
Deadwood abundance	1.21 ^{ab}	1.20 ^{ab}	1.04 ^a	1.0 ^b	12.6	0.014	4; 149
Soil compaction	0.25 ^a	0.70 ^{ab}	0.50 ^{ab}	0.66 ^b	18.6	0.000	3; 137
Soil depth	1.43 ^a	1.20 ^{ab}	1.05 ^{ab}	0.97 ^b	8.99	0.030	3; 137
Disturbance index	0.58 ^a	0.80 ^{ab}	1.45 ^b	2.37 ^c	84.4	<0.001	3; 139
Forest structure variables							
Top height of canopy (m)	31.6 ^c	24.7 ^{bc}	15.1 ^{ab}	9.8 ^a	104.9	<0.001	3; 126
Total seedling density (m ²)	3.4 ^a	5.4 ^a	5.7 ^a	4.3 ^a	66.1	0.010	3; 140
Shrub volume (m ³)	0.14 ^b	0.40 ^b	1.03 ^b	10.1 ^a	81.6	0.041	3; 140
Understorey density (m ²)	0.31 ^b	0.46 ^{ba}	0.34 ^b	0.78 ^a	53.8	0.046	3; 140

H Species list

Table H1: Species recorded in the sample plots: Latin scientific name and abbreviation used in the study. Four species were sampled only at sub-sub plot level. These are: *Arbutus unedo*, *Calicotome villosa*, and *Ulmus minor* in Mazi sacred forest, and *Juglans regia* in Greveniti sacred forest.

Abbreviation	Full species name	Abbreviation	Full species name
abr	<i>Abies borisii-regis</i>	pce	<i>Prunus cerasus</i>
aca	<i>Acer campestre</i>	pco	<i>Prunus cocomilia</i>
amo	<i>Acer monspessulanum</i>	phe	<i>Pinus heldreichii</i>
aop	<i>Acer opalus</i>	pla	<i>Phillyrea latifolia</i>
aps	<i>Acer pseudoplatanum</i>	pma	<i>Prunus mahaleb</i>
car	<i>Colutea arborescens</i>	pni	<i>Pinus nigra</i>
cav	<i>Corylus avellana</i>	psp	<i>Prunus spinosa</i>
cbe	<i>Carpinus betulus</i>	pte	<i>Pistacia terebinthus</i>
cco	<i>Cotinus coggygria</i>	qco	<i>Quercus coccifera</i>
cem	<i>Coronilla emeroides</i>	qde	Deciduous oaks ^a
cma	<i>Cornus mas</i>	qce	<i>Quercus cerris</i> ^b
cor	<i>Carpinus orientalis</i>	qfr	<i>Quercus frainetto</i> ^b
cra	<i>Crataegus sp.</i>	qpe	<i>Quercus petraea</i> ^b
csi	<i>Cercis siliquastrum</i>	qpu	<i>Quercus pubescens</i> ^b
eeu	<i>Euonymus europaeus</i>	qtr	<i>Quercus trojana</i>
fan	<i>Fraxinus angustifolia</i>	rac	<i>Ruscus aculeatus</i>
for	<i>Fraxinus ornus</i>	rca	<i>Rosa canina</i>
fsy	<i>Fagus sylvatica</i>	saa	<i>Salix alba</i>
haz	<i>Corylus colurna</i>	sar	<i>Sorbus aria</i>
jco	<i>Juniperus communis</i>	sdo	<i>Sorbus domestica</i>
jox	<i>Juniperus oxycedrus</i>	sni	<i>Sambucus nigra</i>
jot	<i>Juniperus foetidissima</i>	spu	<i>Salix purpurea</i>
oca	<i>Ostrya carpinifolia</i>	sto	<i>Sorbus torminalis</i>
pav	<i>Prunus avium</i>	til	<i>tilia spp.</i>

Appendix 2 (Chapter IV)

I Tree cores time series

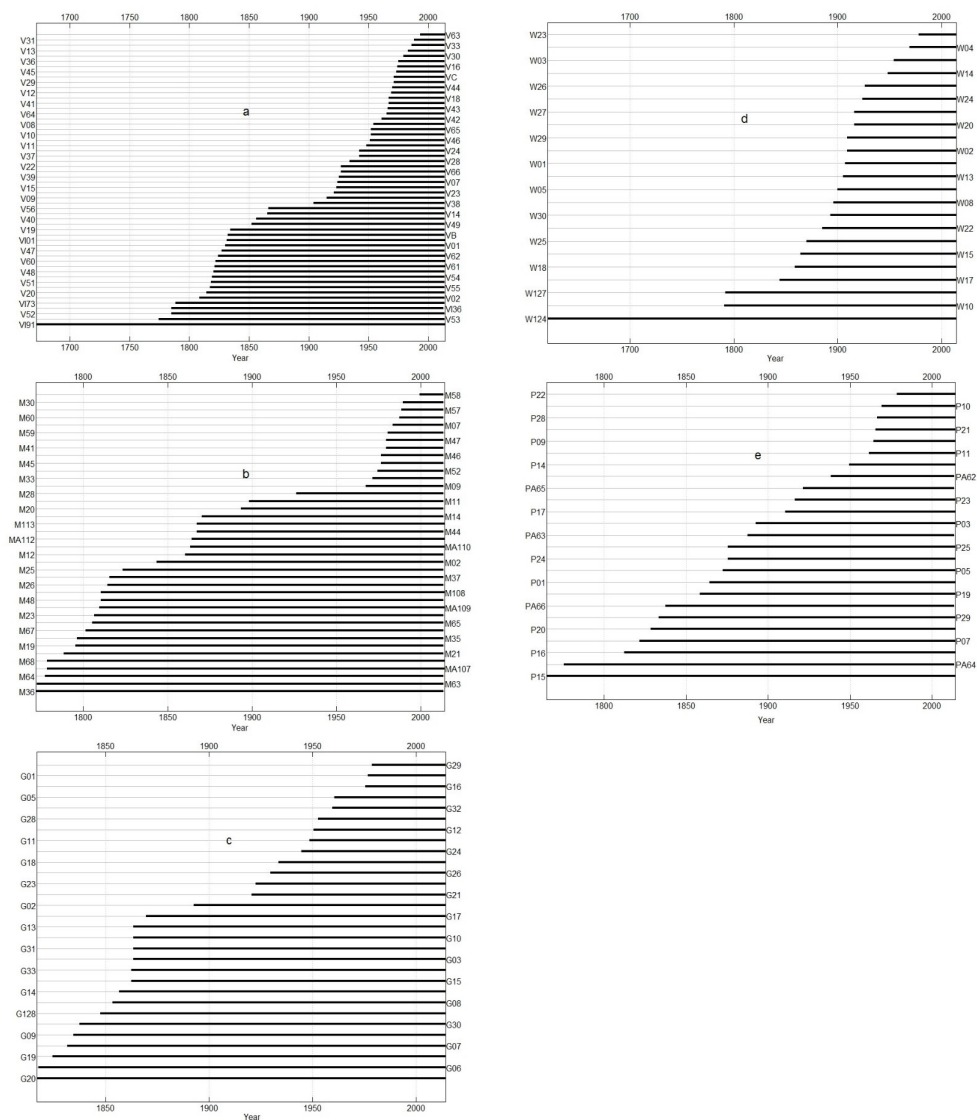


Figure I1: Bar graphs of time spans for all dated tree cores sampled in the five studied sacred forests (a= Vitsa, b= Mazi; c= Greveniti; d= Vovoussa; e= Palioseli). Time series start at 2014 for Vitsa and Mazi; at 2015 for Greveniti, Vovoussa and Palioseli.

J Population data

Table J1: Population data (1812-2015) and relative sources for the five study areas.

Years	Greveniti	Vitsa ^g	Palioseli ^h	Mazi	Vovoussa
1812 ^a	-	1150	-	-	750
1850 ^a	-	940	-	-	-
1856 ^a	-	935	-	-	210
1868 ^a	1500	940	1044	188	210
1874 ^a	-	570	-	-	250
1895 ^b	-	-	1044	-	-
1902 ^a	-	738	-	-	497
1913 ^a	1250	420	-	-	505
1920 ^a	714	317	-	-	-
1928 ^a	676	372	413	-	-
1940 ^a	803	438	458	-	-
1951 ^a	439	181	280	-	-
1961 ^a	473	220	230	-	-
1971 ^a	343	163	153	-	-
1981 ^a	289	140	180	-	-
1986 ^a	255	-	-	-	-
1991	376	154	136	-	-
2000 ^c	-	-	-	-	-
2001 ^d	339	-	112	-	-
2011 ^e	60	102	63	187	102
2015 ^f	50	-	-	-	50

^a Papageorgiou, 1995.

^b Ottoman census, 1895, Thalís SAGE unpublished report.

^c Damianakos et al 1997.

^d Official Greek Census, year 2011.

^e Ethnographic field work (Thalís SAGE unpublished report).

^f Ethnographic field work (Efthimiou, 2019).

^g Vitsa population flows were integrated with data available in <http://www.vitsa.gr/node/213>.

^h Palioseli population flows were integrated with data available in Wikipedia.

K Model fitting

Table K1: Summary of model fitting resultants (linear, log-normal, log-logistic and polynomial) for the tree population of the five sacred forests. For each model the adjusted R^2 and the Akaike Information Criterion (in parenthesis).

Model	Vitsa	Mazi	Palioseli	Vovoussa	Greveniti
Linear	0.08 (406.95)	0.49 (305.12)	0.11 (493.61)	0.12 (668.11)	0.12 (554.56)
Log-normal	0.36 (121.69)	0.56 (106.27)	0.29 (151.53)	0.22 (202.57)	0.39 (168.54)
Log-logistic	0.74 (81.74)	0.62 (98.59)	0.63 (115.96)	0.51 (171.28)	0.80 (113.31)
Polynomial	0.20 (401.80)	0.68 (281.03)	0.22 (487.72)	0.24 (659.08)	0.26 (536.83)

Appendix 3 (Chapter VI)

L Spatial-temporal correlations in environmental phenomena

This appendix provides a short overview of existing correlations between spatial and temporal scales for a selection of environmental phenomena. Data was retrieved from two sources: (i) the International Disaster Database EM-DAT (for worldwide floods, droughts, and wildfires), (ii) the Natural Center for Environmental Information of the National Oceanic and Atmospheric Administration (for worldwide earthquakes, tsunamis, and volcanic eruptions). Observations were organized in a unique panel, identifying the event ID code, the starting year of the event, and the relative intensity. The intensity variable was measured differently, according to the scale in use for the relative phenomenon¹, and then it was uniformed into a 1-4 intensity scale, which was used as a proxy of the spatial scale of the environmental phenomenon. Finally, the average time lag occurring between events was identified for each intensity level and for each environmental phenomenon. Results were plotted as to visually inspect the occurring trend between spatial and temporal scales (Fig. L1).

Results show that four environmental phenomena over six fairly approximate to a linear correlation between temporal and spatial scales (volcanic eruptions, from intensity ≥ 2 ; earthquakes, wildfires, and floods, from intensity ≥ 3).

¹Droughts: Km^2 ; Earthquakes: Richter scale; Floods: Km^2 ; Tsunami: earthquake primary magnitude; Volcanic eruptions: Volcanic Explosivity Index (VEI); Wildfires: Km^2 .

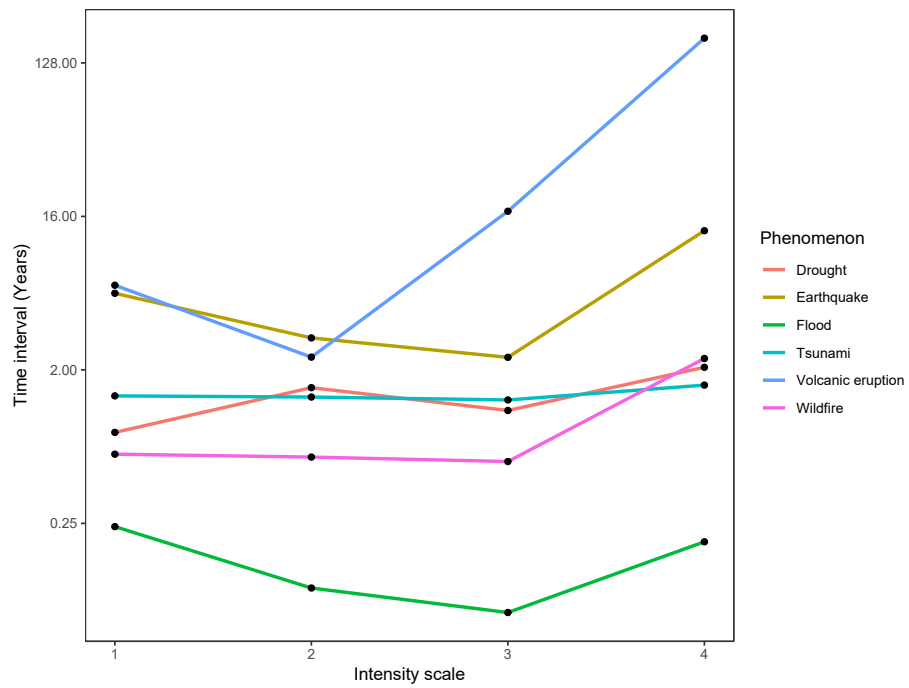


Figure L1: Correlation between time (the frequency of events in time) and space (the intensity of the events) for six global environmental phenomena.

M Social data

In this appendix, the social data which was used to run the simulations is provided. Data was retrieved from on site ethnographic research performed during 2019 in Greveniti village (Efthimiou, 2019). Three main set of data were used to simulate the forest expansion under selected management constraints: (i) village population (P), (ii) main occupation (management) of the villagers (W), (iii) and strength of prohibition rules (F), as any existing cultural taboos prohibiting the access to the forest to people.

Village population data P was retrieved from available census data in the period 1868-2011. As official population statistics do not often match the real population living in villages, available datasets were verified using additional secondary sources, archive data, and ethnographic research. This analysis resulted in a twelve points dataset (Table M1).

Table M1: Population data of the Greveniti village (1868-2015).

Year	Census data ¹	Adjusted census data ²
1868	1500	1500
1913	786	1250
1920	714	714
1928	809	676
1940	830	803
1951	690	439
1961	445	473
1971	591	343
1981	295	289
1991	536	376
2001	162	339
2011	193	60

¹ELSTAT, 2020.

²Census data adjusted after ethnographic research conducted in the field.

To obtain yearly data for the 1868-2015 time period, a non-linear interpolation (spline) was performed to increase the number of observations. The population dataset was smoothed using a Generalized Additive Model (GAM) with years as the model

explanatory variable. This interpolation yielded a R^2 of 0.853 and it is depicted in Fig. M1.

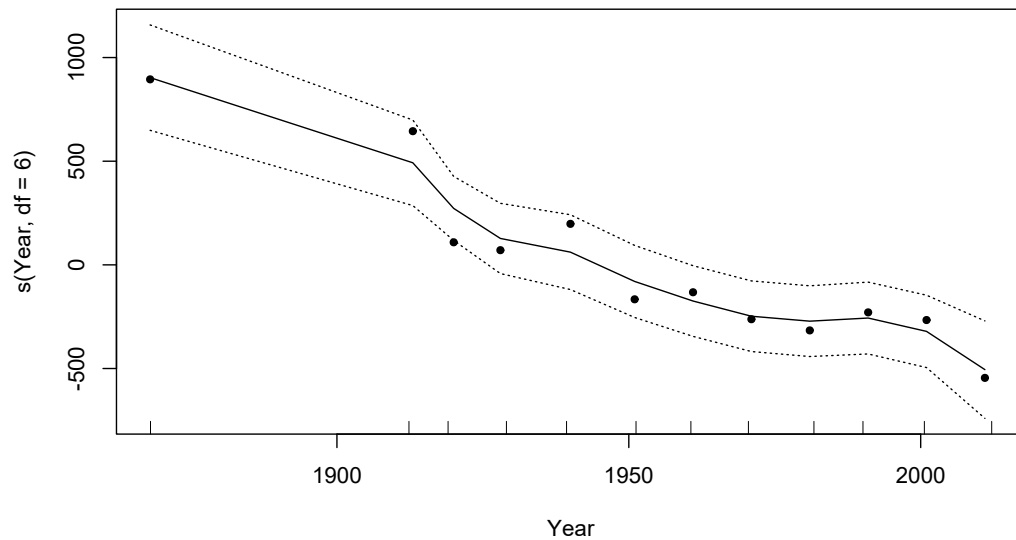


Figure M1: GAM fitting of the Greveniti population dataset . The X axis represents the yearly calendar, while the Y axis is the univariate response variable of the GAM. Black dots indicates the 12-points population dataset, while the black continuous line is its respective GAM interpolation.

Management data of the Greveniti villages was retrieved from existing archive documentations and ethnographic research (Efthimiou, 2019). In particular, we considered four main typologies of management: (i) forestry activities such as woodcutting, (ii) agricultural activities, (iii) pastoralism activities, and (iv) no active management due to abandonment. Using the available qualitative information, we determined whether one (or more) of each management conditions was taking place within the 1868-2015 time interval. Figure M2 provides a summary of the main management changes experienced in Greveniti in the period under consideration.

The strength of prohibition rules in place in Greveniti sacred forest (F) was estimated using information retrieved from interviews conducted in the field (e.g. Efthimiou 2019), and then transformed into a 0-1 scale with 0 meaning no prohibition rules in place, and 1

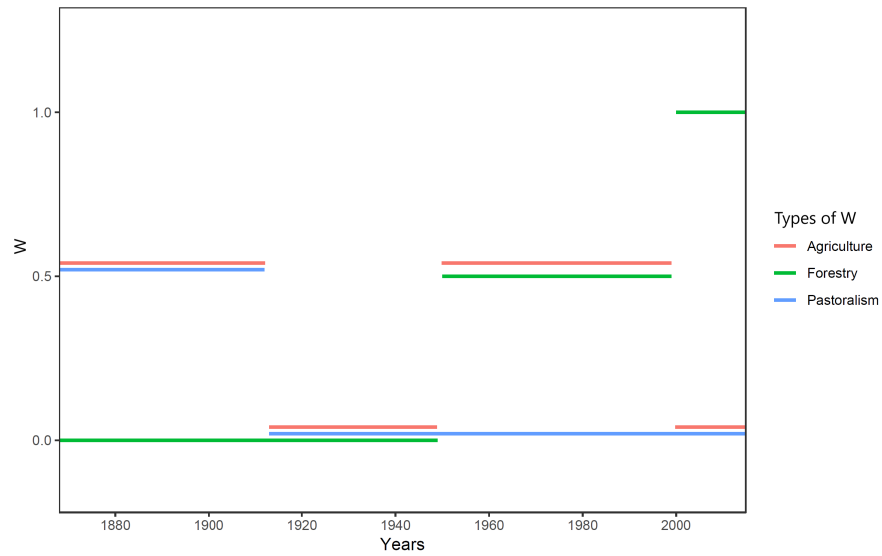


Figure M2: Visualization of the main management changes (W) undergone in Greveniti village from 1868 until 2015. The Y axis indicates which type of management activities was occurring (agriculture, forestry, or pastoralism). ($W = 0$) indicates that no management was taking place.

meaning prohibition rules active. Fig. M3 provides an overview of how prohibition rules have changed in Greveniti sacred forest in the period under consideration.

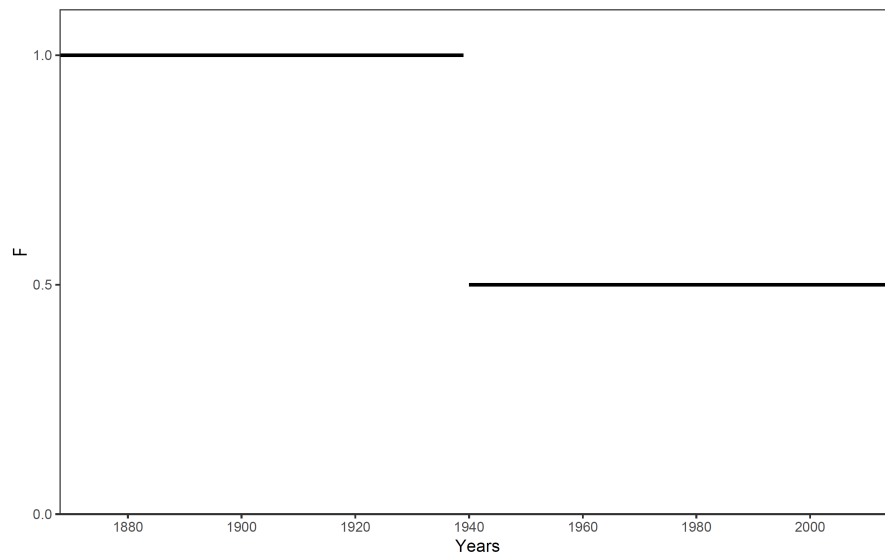


Figure M3: Visualization of the strength of prohibition rules in place in Greveniti sacred forest (F) from 1868 until 2015. The Y axis indicates the strength of prohibition rules. ($F = 0$) indicates that no prohibition rules were in place.

N Model codes

This appendix provides all the programming codes of MOSES. MOSES is made of 11 modules articulated into 13 functions overall. Almost all modules are executed through the main MOSES routine, which calls and executes all related subroutines. The model was coded using Python 2.7 language (Python Software Foundation, 2018).

Main MOSES routine

This routine generates the CA model extent in space and its evolution in time.

```
import pandas as pd
import math
import timeit
import os
import random

# Select the number of simulations to run
def simul (time2):
    sideinnermatrix= 25
    time= 148
    # add population data (P). FOR SCENARIO 1,2, and 3
    pop = []
    with open('fitted_ratio_pop.txt', 'r') as f:
        content = f.readlines()
        for x in content:
            row = x.split()
            pop.append(float(row[3]))
    # make it in percentage
    pop = [x * 100 for x in pop]
    # add management changes data (O). FOR SCENARIO 2, and 3
    manag_data = pd.read_excel('management_greveniti.xlsx')
    # add prohibition rules data (W). FOR SCENARIO 3 only
    know_data = pd.read_excel('knowledge_greveniti.xlsx')

    # Iterate the model for n simulations
    for c in range(time2):
        # store results
        count_cell_with_trees = []
        count_cell_with_know = []
        # Create from management data the initial binary vector (O)
        man= [1.0,0]
        # set timer
        start = timeit.default_timer()
        # import the subroutine creating the initial CA matrix at time 0
        from matrtime0_v2 import trial
        forest=trial(sideinnermatrix)
        import numpy as np
        np.set_printoptions(threshold=np.inf)
        import matplotlib.pyplot as plt
```

```

# import the subroutines creating
# the environmental noises e_k, for K=2,1, and 0 respectively
from spatial_noise_d1 import spanoise
from spatial_noise_d2 import spanoiseee
from spatial_noise_d3 import spanoiseeee
from noisesum import sumnoise
span1=spanoise(sideinnermatrix)
span2=spanoiseee(sideinnermatrix)
span3=spanoiseeee(sideinnermatrix)
noisesum=sumnoise(sideinnermatrix)
#effect of noise on seed survival, time 0,
effect_seeds=noisesum *1.0
effect_seeds = list(effect_seeds.flat)

# Iterate the CA model in time
for v in range(time):
    print(v)
    y=v
    # generate storing vectors
    la = []
    cell_with_trees = []
    cell_with_know = []
    # import the subroutines creating
    # the transitional probabilities of the cell Lefkovitch Matrix
    from inputdata import inputs
    growth = inputs()[1]
    surv= inputs()[0]
    # import the subroutines creating
    # the seeds immigration probabilities, as to integrate the immigration rate I
    from Immatrtime0 import immigration
    finalseeds=immigration(sideinnermatrix, forest)
    # import the subroutines creating the Social layer S
    from social_layer import manag
    man=man
    mana=manag(know_data,manag_data,v,man,pop[v],sideinnermatrix)[1]
    man=manag(know_data,manag_data,v,man,pop[v],sideinnermatrix)[0]
    from matrtime0_v2 import sacred
    ssss =sacred(know_data,sideinnermatrix, v)[1]
    kn_active_cells= len(ssss)
    print(kn_active_cells)

# inner loop across all CA cells
for i in range(len(forest)):
    growth=growth
    surv=surv
    stop = timeit.default_timer()
    # impact of social layer on transitional probabilities
    growthh=[growth[iii]*mana[i][iii] for iii in range(len(growth))]
    survv=[surv[iii]*mana[i][iii] for iii in range(len(surv))]
    #Include the new seeds generated by trees
    a=finalseeds[i]
    cellvalue=(forest[i])
    # add the individual of previous class that did not grow
    # multiply for the survival and NO growth probability

```

```

nuovo1 = [cellvalue[j]*survv[j] for j in range(len(cellvalue))]
#now add the individual of that did grow
# multiply for the survival and growth probability
nuovo2 = [cellvalue[j]*growthh[j] for j in range(1,7)]
# Add the impact of the noise on the seeds that are now growing
# , so to kill them off accordingly
noise1_impact = effect_seeds[0][i] * np.random.normal(0,1.224744871)
noise2_impact = effect_seeds[1][i] * np.random.normal(0,1.224744871)
noise3_impact = effect_seeds[2][i] * np.random.normal(0,0.3)
#make the combined effect
comb_effect_noise= np.exp(noise1_impact+noise2_impact+noise3_impact)
# multiply for seed survival baseline (0.00000095)
all_effect_seeds=comb_effect_noise*0.00000095
#finally multiply for seeds to generate first year seedlings
nuovo2.insert(0, cellvalue[0]*all_effect_seeds)
#insert seeds value, after the effect of the noise
nuovo2.insert(0, a)
#print(nuovo2)
nuovo2.pop()
# add the two components together
last = [nuovo1[j]+ nuovo2[j] for j in range(len(nuovo2))]
# Insert a maximum carrying capacity for the cell.
# If cell reach a certain threshold of Basal Area, trees die
bacost=0.00007854
basareacell= ((last[2]*(2.5*2.5)*bacost)+(last[3]*(7.5*7.5)*bacost)+
(last[4]*(20*20)*bacost)+( last[5]*(40*40)*bacost)+(last[6]*(75*75)*bacost))
basal2= basareacell/(15*15)*100
if basal2>=100:
lastshed=[last[i]-(last[i]*0.5) for i in range(len(last))]
else:
lastshed=last

# import the subroutines creating the areas with the prohibition rules (W)
from matrtime0_v2 import sacred
sacred =sacred(know_data,sideinnermatrix, 0)[0]
sasa = sacred[i]
if sasa != 1.0 and
(lastshed[3]+lastshed[4]+lastshed[5]+lastshed[6])>=1:
cell_with_trees.append(1)
# and now give me the cell WITHIN the sacred forest,
that have active management
if sasa == 1.0:
if (mana[i] ==[0,0,0,0,0,0,0] or mana[i]==[0,0,0.1,0.5,1,1,1]
or mana[i]==[1,1,1,1,0.5,0.1,0.1]):
cell_with_know.append(1)
# store results of CA for the next step in the temporal loop
la.append(lastshed)
forest=la #Execute loop
man=man #Execute loop
# count cells which have been colonized
coun=(cell_with_trees.count(1))
count_cell_with_trees.append(coun)
# count cells with active management within the sacred forest
kno=(cell_with_know.count(1))

```

```

count_cell_with_know.append(kno)

# Correlate the spatial noise in time (K=0)
for i in range(sideinnermatrix*9):
    a1=random.random()
    #this is the time interval for the first scale noise, it is equal to 1 year
    t_int=sideinnermatrix/sideinnermatrix
    position=((v+1))/float(t_int)
    expon=math.exp(-position)
    #check if it is larger then a distribution based on time
    if (a1 < expon):
        span1[i,:,:]=span1[i,:,:]
    else:
        if (a1 < 0.5):
            span1[i,:,:]=1-span1[i,:,:]
        else:
            span1[i,:,:]= span1[i,:,:]
    span1= span1.reshape(15,15,int(math.sqrt(sideinnermatrix)),
int(math.sqrt(sideinnermatrix))
    .swapaxes(1,2).reshape(sideinnermatrix*3,sideinnermatrix*3)
# Correlate the spatial noise in time (K=1)
for i in range(9):
    #this is the time interval for the second scale noise, it is equal to 25
    t_int_v2=sideinnermatrix
    delta_time_v2=(v+1)/t_int_v2
    a2=random.random()
    aa2=random.random()
    #check if it is larger then a distribution based on time
    if (((v+1)-(t_int_v2*delta_time_v2)) % float(t_int_v2))==0)
    position_v2=(v+1)/float(t_int_v2)
    expon_v2=math.exp(-position_v2)
    else:
        position_v2=((v+1)-(t_int_v2*delta_time_v2))/float(t_int_v2)
        expon_v2=math.exp(-position_v2)
    if (a2 < expon_v2):
        span2[i,:,:]=span2[i,:,:]
    else:
        if (aa2 < 0.5):
            span2[i,:,:]=1-span2[i,:,:]
        else:
            span2[2,:,:]= span2[i,:,:]
    span2= span2.reshape(3,3,int((sideinnermatrix)),int((sideinnermatrix)))
    .swapaxes(1,2).reshape(sideinnermatrix*3,(sideinnermatrix*3))
    noise_due=span2[1,1]
    save_noise2.append(noise_due)

# Correlate the spatial noise in time (K=2)
a3=random.random() # extract a random number between 0 and 1
aa3=random.random() # extract a random number between 0 and 1
#this is the time interval for the third scale noise, it is equal to 625
t_int_v3=sideinnermatrix*sideinnermatrix
delta_time_v3=(v+1)/t_int_v3
#check if it is larger then a distribution based on time
if (((v+1)-(t_int_v3*delta_time_v3)) % float(t_int_v3))==0):

```

```

position_v3=(v+1)/float(t_int_v3)
expon_v3=math.exp(-position_v3)
else:
position_v3=((v+1)-(t_int_v3*delta_time_v3))/float(t_int_v3)
expon_v3=math.exp(-position_v3)
#check if it is larger then a distribution based on time
if (a3 < expon_v3):
span3=span3
else:
if (aa3 < 0.5):
span3=1-span3
else:
span3=span3
noise_tre=span3[1,1]
save_noise3.append(noise_tre)
# sum all noises processes
noisesum= np.add(span1,span2)
noisesum= np.add(noisesum,span3)
# impact seeds accordingly
effect_seeds=noisesum*1.0
#reshape it as 2d
effect_seeds = list(effect_seeds.flat)
span1 = np.lib.stride_tricks.as_strided(span1, shape = (15, 15, 5, 5),
strides = (3000,40,600,8))
span1=span1.reshape(225,5,5)
span2 = np.lib.stride_tricks.as_strided(span2, shape = (3, 3, 25, 25),
strides = (15000,200,600,8))
span2=span2.reshape(9,25,25)
count_cell_with_trees=str(count_cell_with_trees)

# Write data to file
for r in RESULTS:
resultFyle.write(count_cell_with_trees + "\n")
resultFyle.close()
count_cell_with_know=str(count_cell_with_know)
for t in RESULTS_2:
resultFyle_2.write(count_cell_with_know + "\n")
resultFyle_2.close()
stop = timeit.default_timer()
print (stop - start)

#-----

```

Input_data

This routine generates the transitional population probabilities for each cell.

```

import math
import timeit
import numbers
import Tkinter
import tkFileDialog
import os

```



```

import random
from random import randint
def inputs ():
    import random
    seeds = random.sample(range(1,9999),8)
    # Fecundity rates
    seeds[0]=0
    seeds[1]=0
    seeds[2]=0
    seeds[3]=9651.597472
    seeds[4]=29286.40581
    seeds[5]=172970.9122
    seeds[6]=3870439.877
    seeds[7]= 0 # just to make the loop working
    # Survival rates
    SUR=[0.00000095, 0.10000, 0.65000,0.88000,0.95200, 0.97800, 0.98780]
    # Growth rates
    gro= [1,1, 0.250986538461538, 0.263750769230769,0.0658918269230769,
    0.0644088995726496, 0.0211039100609967]
    sur= [1-gro[i] for i in range(len(gro))] # this should be 1-growth
    la = []
    cell_with_trees = []
    # vector survival and growth
    growth=[gro[iii]*SUR[iii] for iii in range(len(gro))]
    # vettore survival and NOT growth
    surv=[sur[iii]*SUR[iii] for iii in range(len(sur))]
    #fertility rates
    fer = [seeds[f]*surv[f] +seeds[f+1]*growth[f] for f in range(len(surv))]
    return (surv, growth,fer)
#-----

```

Crea_cell_time_0

This routine generates two functions: (i) the seeds rain which enters all cells of the CA, and (ii) the structure of the initial *i* cell of the CA model.

```

# function which creates the seed reain entering each year in the CA model
def seedrain():
    import struct
    import random
    seedrain= random.randint(0,300)
    return (seedrain)

# function which creates the generic strucure of
the initial CA cell
def celzero():
    import random
    cell0 = [0] *7
    cell0[0]=seedrain()
    cell0[1]=np.random.normal(37.5,133.2)
    cell0[2]=np.random.normal(24.3,47.2)
    cell0[3]=np.random.normal(7.7,10.2)

```

```

cell0[4]=np.random.normal(1.9,3.7)
cell0[5]=np.random.normal(0.9,1.4)
cell0[6]=np.random.normal(0.7,0.7)
for i in range(len(cell0)):
    if cell0[i] < 0:
        cell0[i]=0
return(cell0)
#-----

```

Matr.time_0

This routine generates the initial Lattice in which the CA is set, as well as the portion of the lattice in which the prohibition rules are existing.

```

import numpy as np
import math
import matplotlib as ml
import matplotlib.pyplot as plt
import random
# make CA at time zero
def trial (sideinnermatrix):
    innermatrix=[]

# import subroutines
from creacelltime0 import celzero
from creacelltime0 import seedrain

matr3d_25x25=np.zeros((9,int(sideinnermatrix),int(sideinnermatrix)))
# generate a forest within the center of the CA lattice
matr3d_25x25[4,:,:].fill(1)
#reshape it as 2d
matr3d_25x25_sh= matr3d_25x25.reshape(3,3,int(sideinnermatrix),
int(sideinnermatrix)).swapaxes(1,2).
reshape(sideinnermatrix*(sideinnermatrix*9))
matr3d_25x25_sh=np.array(matr3d_25x25_sh).tolist()
#populate it with the cells created in the subroutine
for i, item in enumerate( matr3d_25x25_sh):
    if (item == 1):matr3d_25x25_sh[i] = celzero()
    else:
        a=seedrain()
        matr3d_25x25_sh[i] = [a,0,0,0,0,0,0]
return(matr3d_25x25_sh)

# function which creates the spatial extent of the lattice
with prohibition rules.
def sacred (know_data,sideinnermatrix, time):
    sacred=np.zeros((9,int(sideinnermatrix),int(sideinnermatrix)))
# number of cell of main sacred forest with active prohibition rules (W)
know_sg = [1]*(int(know_data['knowledge'][time]*(sideinnermatrix*sideinnermatrix))
no_know_sg=[0]*(int((sideinnermatrix*sideinnermatrix)-len(know_sg)))
# now shuffle them together
knowledge=(know_sg+no_know_sg)

```

```

know_shuffled = random.sample(knowledge, len(knowledge))
know_shuffled_n = np.array(know_shuffled).reshape(25, 25)
#fill the core with 1 or 0 depending to
# the above reshuffling (, i.e. the sacred forest)
sacred[4,:,:]= know_shuffled_n
#reshape it as 2d
sacred= sacred.reshape(3,3,int(sideinnermatrix),int(sideinnermatrix))
.swapaxes(1,2).reshape(sideinnermatrix*(sideinnermatrix*9))
return(sacred, know_sg)
#-----

```

Emi_matr_time.0

This routine generates the number of seeds which emigrates from each cell to its neighbours.

```

import math
import numpy
import Tkinter
import tkFileDialog
import os
def emigration (sideinnermatrix, forestzero):
# import the subroutines generating the seedrain,
#the cell structure, and the inputs data
from creacelltime0 import seedrain
from inputdata import inputs
fer=inputs()[2]
seedrain =seedrain()
emigra = []
totseeds = []
seee= []

for i in range(len((forestzero))):
#definition of variables:
cell=forestzero[i]
# emigration vector to neighbouring cells.
emi= [0.4, 0.4, 0.4, 0.4, 0.4, 0.4, 0.4]
#seed produced for each size classes minus the emigrated seeds
see = [fer[j]*round(cell[j])*(1-emi[j]) for j in range(len(cell))]
#number of seeds produced by each tree class
totsum= sum(see)
totemi= sum([fer[j]*round(cell[j])*(emi[j]) for j in range(len(cell))])
emigra.append(totemi)
totseeds.append(totsum)
#arrange the array into a matrix
import numpy as np
import math
totseedsmatrix=np.reshape(totseeds, (sideinnermatrix*3,sideinnermatrix*3))
return(emigra,totseedsmatrix)
#-----

```

Im_matr_time_0

This routine generates the number of seeds which immigrate to each cell from its neighbours.

```
import numpy as np
import math
def immigration (sideinnermatrix, forestzero):
from emimatrttime0 import emigration
#choose only first of the two function
outputsemigra=emigration(sideinnermatrix, forestzero)[0]
#individual emigration: divide all emigration seeds by neighbourin receivers
# (8) to know the individual seeds each cell will receive
indemi = [emigra[i]/8 for i in range(len(emigra))]
immigra = []
immatrix=np.reshape(indemi, (sideinnermatrix*3,sideinnermatrix*3))
# compute the value of immigration from neighbouring cells
immatrix2 = np.zeros((sideinnermatrix*3,sideinnermatrix*3))
for i in range(sideinnermatrix*3): #rows
for j in range(sideinnermatrix*3): #columns
if i==0 or i==sideinnermatrix*3-1 or j==0
or j==sideinnermatrix*3-1: immatrix2[i,j]=0
else:
# Moore neighbourood
immatrix2[i,j] =immatrix[i,j+1] + immatrix[i,j-1] + immatrix[i+1,j]
+ immatrix[i-1,j] + immatrix[i-1,j+1]+ immatrix[i-1,j-1]
+ immatrix[i+1,j+1] +immatrix[i+1,j-1]
# now move the immigration value on the total seeds
totseedsmatrix=emigration(sideinnermatrix, forestzero)[1]
#add immigrations
finalseeds=totseedsmatrix+immatrix2
# as list
import numpy as np
finalseeds=np.array(finalseeds).tolist()
seeds= []
for i in range(len(finalseeds)):
f=finalseeds[i]
for j in range(len(f)):
g=f[j]
seeds.append(g)
return(seeds)
#-----
```

Social_layer

This routine generates the social layer S which impacts the vegetation of each cell of the CA.

```
import timeit
import random
import numpy as np
```

```

# define management function
def occupation (manag_data,time):

none = [1] *148
occupation = np.array([manag_data['woodcutting'][time],manag_data['grazing']
[time],manag_data['agriculture']
[time],manag_data['none_of_the_three'][time],none[time]])
return(occupation)

def manag (know_data,manag_data,time,man, pop,sideinnermatrix):

#management change iteratively accoridngly to population ratio
(e.g. if pop decreases prob of management also decreases)
# impact of population on management
if (man[-1]==0 and pop<0.0):
mana_pop=[1,0]
elif (man[-1]>=1 and pop>0.0):
mana_pop=[0,1]
elif (man[-1]>=1 and pop<0):
mana_pop=[-(pop/100),(1+(pop/100))]
else:      # altrimenti fai la solita roba
mana_pop = [man[i]*((sum(man)-man[1])-(pop/100))
/sum(man[:1]) for i in range(len(man))]
#sum sliced up to the three first values of man list, so to exclude "none"

# now add the value for no active management
mana_pop[1] = pop/100+man[1]

# SCENARIO 2; double effect of management changes and population decline
mana_pop_wg = np.array([mana_pop[0],mana_pop[0],
mana_pop[0],mana_pop[0],mana_pop[1]])
from social_layer import occupation
occupation=occupation(manag_data,time)
mana_final = mana_pop_wg*occupation
# impact on survival in % from woodcutting,
# higher more survival, lower less survival
woodcut=[1,1,1,1,0.5,0.1,0.1]
# impact on survival from grazing, higher more survival, lower more dead
grazing=[0,0,0.1,0.5,1,1,1]
# impact on survival from agriculture all dead
agri=[0,0,0,0,0,0,0]
# impact on survival from no management, higher more survival, lower more dead
none=[1,1,1,1,1,1,1]
nwoodcut=[woodcut]*((int(mana_final[0]*(125*125)))
ngrazing=[grazing]*((int(mana_final[1]*(125*125)))
nagri=[agri]*((int(mana_final[2]*(125*125)))
nnone=[none]*((125*125)-(len(ngrazing)))
social=(nwoodcut+ngrazing+nagri+nnone)
shuffled = random.sample(social, len(social))
# SCENARIO 3 - Prohibition rules
from matrtime0_v2 import sacred
sacred=sacred(know_data,sideinnermatrix, time)[0]
final_manag=list(range(0,(25*sideinnermatrix*9)))

```

```

for i, item in enumerate(sacred):
if (item == 1):final_manag[i] = none
else:
final_manag[i] = shuffled[i] # if not, the shuffled vector

mana_final =[(mana_final[0]+mana_final[1]+mana_final[2]
+mana_final[3]),mana_final[4]]
return (mana_final, final_manag)
#-----

```

spatial_noise_d1

This routine generates the environmental noise which impacts the CA at its finer spatial scale (5x5 cells).

```

import math
import timeit
import random
import Tkinter
import tkFileDialog
import numpy as np
from random import randint
def spanoise (sideinnermatrix):
matr3d_5x5=np.random.random((int(sideinnermatrix*9),
int(math.sqrt(sideinnermatrix)),int(math.sqrt(sideinnermatrix))))
random_noise=np.random.randint(2, size=sideinnermatrix*9)
#now populate the matrix with random numbers.
# Each 3 dimension matrix will have same numbers,
# but all matrices will be different
for i in range(sideinnermatrix*9):
matr3d_5x5[i,:,:].fill(random_noise[i])
return(matr3d_5x5)
#-----

```

spatial_noise_d2

This routine generates the environmental noise which impacts the CA at its medium spatial scale (25x25 cells).

```

import math
import timeit
import random
import Tkinter
import tkFileDialog
import numpy as np
from random import randint
np.set_printoptions(threshold=np.inf)
# NOISE2 same of before but this time for a 25x25 cells matrix.
def spanoisee (sideinnermatrix):
matr3d_25x25=np.zeros((9,int(sideinnermatrix),int(sideinnermatrix)))

```

```

random_noise25x25=np.random.randint(2, size=9)
for i in range(9):
    matr3d_25x25[i,:,:].fill(random_noise25x25[i])
return(matr3d_25x25)
#-----

```

spatial_noise_d3

This routine generates the environmental noise which impacts the CA at its coarser spatial scale (75x75 cells).

```

import math
import timeit
import random
import Tkinter
import tkinterFileDialog
import numpy as np
from random import randint
# NOISE3 same of before but this time for the whole lattice.
def spanoiseeee (sideinnermatrix):
    a=random.randint(0,1)
    matr3d_125x125=np.full((((int(sideinnermatrix*3),int(sideinnermatrix*3))), int(a))
    return(matr3d_125x125)
#-----

```

Noise_sum

This routine sums the three environmental noises into a unique layer.

```

import math
import timeit
import random
import Tkinter
import tkinterFileDialog
import numpy as np
from random import randint
np.set_printoptions(threshold=np.inf)
def sumnoise(sideinnermatrix):
    from spatial_noise_d1 import spanoise
    from spatial_noise_d2 import spanoiseee
    from spatial_noise_d3 import spanoiseeee
    noise1=spanoise(sideinnermatrix)
    noise1= noise1.reshape(15,15,int(math.sqrt(sideinnermatrix)),
    ,int(math.sqrt(sideinnermatrix))).swapaxes(1,2)
    .reshape(sideinnermatrix*3,(sideinnermatrix*3))
    noise2=spanoiseee(sideinnermatrix)
    noise2= noise2.reshape(3,3,int((sideinnermatrix)),
    int((sideinnermatrix))).swapaxes(1,2).
    reshape(sideinnermatrix*3,(sideinnermatrix*3))
    noise3=spanoiseeee(sideinnermatrix)
    noisesum= np.add(noise1,noise2)

```

```
return(noisesumsum)
```

```
#-----
```


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