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"RESTAURACIÓN ECOLÓGICA DE BOSQUES TEMPLADOS BAJO DIFERENTES CONDICIONES DE DISTURBIO: DESARROLLO DEL DOSEL Y SOTOBOSQUE".

T E S I S

QUE PARA OBTENER EL GRADO ACADÉMICO DE

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INDICE

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Resumen

En el presente estudio se aborda la importancia de la Ecología de la Restauración para recuperar diversos componentes de ecosistemas templados bajo diferentes condiciones de perturbación. A través de experimentos de restauración y rehabilitación ecológica en tres sitios de la Comunidad Indígena de Nuevo San Juan Parangaricutiro (CINSJP), Michoacán, México que diferían en la identidad de los agentes causantes de perturbación se evaluaron las barreras a la restauración. Los sitios de estudio fueron: un arenal remanente de la erupción del Volcán Paricutín (Mesa de Cutzato) con una capa de ceniza volcánica de entre 30 y 70 cm de profundidad; un arenal remanente del mismo evento pero al cual se le extrajo la arena con fines de aprovechamiento (Llano de Pario); y un campo agrícola abandonado (San Nicolás) por los escasos rendimientos de los cultivos. En los tres sitios se realizaron diversas acciones de restauración o rehabilitación ecológica (plantado de especies arbóreas nativas y exóticas, sembrado de leguminosas arbustivas, uso de acolchados orgánicos) y se evaluaron durante cuatro años. Se encontró que en los arenales de Cutzato y Pario la recuperación de la cobertura arbórea es difícil y requieren de varias acciones de rehabilitación concertadas, y que su eficacia depende de la variabilidad climática interanual. En San Nicolás, se encontró que las condiciones del sitio permiten la restauración ecológica en sentido estricto. El sembrado de leguminosas incrementó la supervivencia y crecimiento de *Abies religiosa* mediantes interacciones de tipo nodriza, mientras que para *Pinus montezumae* y *P. pseudostrobus* no tuvo un efecto. Además, las leguminosas arbustivas incrementaron la biodisponibilidad de nutrientes esenciales para las plantas y facilitaron el establecimiento de especies nativas incrementando la diversidad.

Palabras clave: Restauración ecológica, coníferas, Lupinus elegans, diversidad, vulcanismo

Abstract

The potential of ecological restoration to recover several components of temperate ecosystems with different degrees of disturbance is explored. Through restoration and rehabilitation experiments that were carried out in three different sites within the Comunidad Indigena de Nuevo San Juan Parangaricutiro, Michoaćan, México that differed in the disturbance agents and also in the level of disturbance restoration barriers were evaluated.

The sites were: a sand deposit remnant of the Paricutin Volcano eruption with a layer of volcanic ash between 30 and 70 cm deep (Mesa de Cutzato); a sand deposit from the mase event but in which the sand was removed with commercial purposes (Llano de Pario); and an abandoned agricultural field (San Nicolas) that was left fallow because of low productivity. In the three sites several restoration or rehabilitation efforts were carried out (planting of tree species, native and exotic, seedling of legume shrubs, addition of organic mulching) and were followed for 4 consecutive years. It was found that in sand deposits of Cutzato and Pario, tree canopy cover recovery is difficult because several rehabilitation actions are needed, and their efficacy varies in response to inter annual weather variability.

In San Nicolas, site conditions allow for the implementation of restoration goals in a strict sense. Legumes increased survival and growth of *Abies religiosa* through nurse plant interactions, but for *Pinus montezumae* and *P. pseudostrobus* there was no effect. Similarly, seeding of shrub legumes increased nutrient bioavailability for plants and increased understory plant species richness.

Keywords: Ecological restoration, conifers, Lupinus elegans, diversity, volcanism

INTRODUCCION.

El Estado de Michoacán cuenta con numerosas masas forestales, muchas de las cuales presentan diferentes tipos de perturbación, principalmente los bosques que se ubican en el Sistema Volcánico Transversal, que durante mucho tiempo han sufrido los efectos de la explotación forestal inadecuada y cambio de uso del suelo, ya sea para agricultura, ganadería, fruticultura o para el establecimiento de asentamientos humanos. La madera de los pinos (*Pinus* sp) es la base de la industria forestal en Michoacán, con aproximadamente 1.2 millones de metros cúbicos en rollo anuales de producción legal, además de la resina, que lo hace el primer productor a nivel nacional con aproximadamente 25 mil toneladas al año (Madrigal 1997).

Sin embargo, el aprovechamiento forestal es la razón por la cual se pierden entre 40 000 y 50 000 has promedio de bosque cada año, dejando alrededor de 704 000 ha de bosque perturbado (Magaña y Madrigal 2000; Masera 1995), aunque la COFOM (2001) estima 35,000 ha por año.

A lo anterior hay que agregar las perturbaciones naturales que, aunque muy poco frecuentes, pueden afectar grandes extensiones boscosas, como lo fue la erupción del volcán Paricutín el siglo pasado.

Debido a la extensión y gravedad de los problemas asociados con la degradación de los bosques, la restauración de estos ecosistemas es deseable para prevenir pérdidas mayores de biodiversidad y para restablecer algunos de los recursos y servicios que proporcionan, como la producción de madera y celulósicos, la captura de carbono, la conservación del suelo y del agua, y para mejorar la calidad de vida de los habitantes tanto de zonas rurales como urbanas (Gregory e Ingram 2000, Masera 1995). Los niveles de destrucción de la cubierta vegetal, del suelo fértil y de la capacidad de regeneración de la vegetación nativa marcarán la pauta del origen y las características biológicas de las especies que podrán usarse para cada localidad ya que los ecosistemas son dinámicos y pueden ya no ser totalmente adecuados para las especies que originalmente existían o las que se piensa deberían existir en dichos ecosistemas.

El concepto de restauración ecológica se entiende como el proceso de asistir en la recuperación de un ecosistema que ha sido degradado, dañado o destruido (SER 2004). El término restauración aún no es bien entendido y se ha aplicado en forma errónea para diversas actividades de reforestación o de regeneración natural, conceptos que aunque presentan similitud, difieren en la metodología aplicable y las metas que persiguen en cada caso. No obstante, la restauración tiene una amplia aplicación para el rescate de diversas áreas afectadas por causas de orden natural (huracanes, tormentas eléctricas, incendios, inundaciones, derrumbes y otros) o antropogénica (contaminación, tala, quema y otros) y para especies que se encuentren en algún grado de vulnerabilidad.

La reforestación es el principal esfuerzo realizado para aminorar la tendencia negativa en la destrucción de los recursos forestales en el Estado de Michoacán. La mayor desventaja de la reforestación como técnica, es que la supervivencia de las plantas suele ser demasiado baja, lo que implica una pérdida importante de recursos económicos y humanos. En Michoacán particularmente, la supervivencia se ha calculado en 37.8% para el primer año de plantación (Sáenz-Romero y Lindig-Cisneros 2004). La alta mortalidad, según reportes de la

SEMARNAP (2000), se debe principalmente a la mala elección de especies (17%), sequía(15%), heladas (14%), fauna nociva (12%), pastoreo (11%), fechas inadecuadas de plantación (8%), competencia vegetal (7%), incendios (6%), vandalismo (5%) y pobre calidad de la planta (5%) (Sáenz-Romero y Lindig-Cisneros 2004).

Como es evidente, 46% de la mortalidad registrada para las reforestaciones en el Estado de Michoacán se presenta por condiciones ambientales adversas para las especies utilizadas (inadecuada selección de especies, sequías y heladas). Esto está ligado a la falta de información de la respuesta fisiológica de estas especies con respecto a los sitios y condiciones donde están siendo utilizadas para reforestación.

Zobel y Talbert (1999) recomiendan la elaboración de experimentos de prueba, denominados “ensayos de especies”, (“zoneamiento ecológico”, Golfari *et al* 1978) para poder determinar de mejor manera el potencial de las especies para adaptarse a las condiciones de los sitios de reforestación. Estos ensayos permiten reducir de manera importante los efectos negativos sobre la supervivencia y desarrollo de las especies elegidas para una reforestación, determinando a priori qué especies presentan mejores características para soportar y en dado caso, aclimatarse a su nuevo ambiente.

La Comunidad Indígena de Nuevo San Juan Parangaricutiro (CINSJP) ha aplicado un modelo forestal sostenible reconocido a nivel mundial. En un esfuerzo por continuar con estrategias de buen aprovechamiento, la comunidad está apoyando los estudios que permitirán la restauración de sitios con diferente nivel de degradación (así como distintos agentes de perturbación), tales como depósitos de ceniza volcánica remanentes de la erupción del Paricutin (arenales) y campos agrícolas abandonados. En respuesta a esta necesidad en 2004 se propusieron una serie de experimentos con la finalidad de evaluar las acciones necesarias para recuperar la cobertura vegetal nativa y recuperación de la biodiversidad en estos sitios. En los sitios cubiertos por arena volcánica los comuneros han iniciado la reforestación y rehabilitación colectiva de esas áreas, acción considerada dentro del Programa de manejo forestal sustentable para el aprovechamiento de los recursos forestales maderables con carácter de persistente en los bosques de la Comunidad Indígena de Nuevo San Juan Parangaricutiro. Dichas prácticas de manejo sumadas a las técnicas desarrolladas en este estudio están encaminadas a recuperar cobertura arbórea con fines de manejo forestal (rehabilitación ecológica con fines productivos). También se han estado aprovechando los arenales para establecer huertos frutícolas con un sistema intensivo y sofisticado, actividades que claramente no pueden ser consideradas como de restauración o rehabilitación ecológica.

Por otra parte la CINSJP prevé la reforestación de las áreas agrícolas que potencialmente son forestales y que en el pasado fueron desmontadas para el uso agrícola de autoconsumo abandonadas. Aun cuando la comunidad de Nuevo San Juan ha reservado tradicionalmente pocas áreas para fines conservación si está interesada en incrementar dicha superficie y es en uno de los sitios del presente estudio (campo agrícola abandonado) donde se podrían intentar escenarios de restauración ecológica en sentido estricto (recuperación de cobertura arbórea nativa, recuperación de biodiversidad y de servicios ambientales) debido al menor nivel de degradación que presenta en comparación con los arenales.

La presente tesis consta de cinco capítulos; siendo el primero una revisión bibliográfica acerca de un tema importante en Ecología de la Restauración: aspectos conceptuales de esta disciplina. Este primer capítulo contiene una breve revisión conceptual de la Ecología de la Restauración, comenzando con una clarificación de la terminología usada en Restauración Ecológica, abordando también el concepto de los estados, transiciones y umbrales en Ecología de Restauración y por último se revisan los conceptos teóricos más recientes desarrollados para la Ecología de la Restauración.

Los siguientes cuatro capítulos de la tesis son el resultado de experimentos de campo en tres sitios de la Comunidad Indígena de Nuevo San Juan Parangaricutiro en el estado de Michoacán para probar la efectividad de varias técnicas que buscan restaurar algunos componentes (estructura, composición y función) de tres ecosistemas degradados; un campo agrícola abandonado, un arenal volcánico remanente de la erupción del Volcán Paricutín (ocurrida en 1943-1952) y por último un arenal volcánico sujeto a manejo (extracción de la totalidad de la capa de arena para ser usada en la industria de la construcción).

El capítulo 2 "Native pine species performance in response to age at planting and mulching in a site affected by volcanic ash deposition" publicado en 2008 en ***New forest*** se hace una evaluación de las barreras a la restauración en el arenal volcánico remanente de la erupción del Volcán Paricutín usando dos especies arbóreas y tratamientos de cobertura.

En el capítulo 3; “Restauración de cobertura forestal en presencia de condiciones ambientales estresantes: Resistencia a bajas temperaturas de tres especies de coníferas”, se hace una evaluación de las barreras a la restauración que presenta un arenal volcánico degradado, se mide el desempeño de tres especies arbóreas ante una condición ambiental severa (bajas temperaturas) y se enumeran las barreras para el establecimiento de leguminosas y pinos.

En el capítulo 4; “Restauración de cobertura forestal en un campo agrícola abandonado a través de la interacción nodriza- coníferas”, se evalúan las barreras a la restauración de un campo agrícola abandonado y se evalúa la magnitud y signo del nodricismo para especies tolerantes e intolerantes a la sombra (*Abies religiosa, Pinus montezumae* y *P. pseudostrobus* de dos edades (7 y 17 meses).

Finalmente en el capítulo 5; “Restauración de la riqueza de plantas vasculares en un campo agrícola abandonado mediante el uso de plantas nodriza”, se evalúa si el uso de leguminosas puede incrementar la riqueza florística en reforestaciones mediante una evaluación de la riqueza de especies vasculares durante tres años posteriores al establecimiento de en un proyecto de restauración ecológica.

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CAPITULO 1. REVISIÓN CONCEPTUAL DE LA RESTAURACIÓN ECOLÓGICA.

La Restauración Ecológica es una actividad intencional que inicia o acelera la recuperación de un ecosistema con respecto a su salud, integridad y sustentabilidad (SER 2004). Ha sido practicada durante décadas al menos en sus formas más conocidas y aplicadas (control de erosión, reforestación y mejoramiento de hábitats). En cambio, la Ecología de la Restauración es una rama de la ciencia asociada tanto con la ecología como con la restauración ecológica, constituye un campo académico y científico muy joven, y es en los últimos 15 años cuando se ha convertido en una disciplina importante, atrayendo investigación básica y con una creciente aparición de revistas indizadas especializadas sobre el tema (Young *et al.* 2005).

La ecología de la restauración comienza a cobrar importancia en nuestro país como disciplina científica y es posible que cobre mayor relevancia con el paso del tiempo. Debido a que es una disciplina reciente, existe una gran variedad de términos - muchas veces similares o cuando menos muy relacionados entre sí- que se utilizan de manera confusa o poco clara en ámbitos diversos, incluido el científico y académico. Hobbs y Norton (1996) atribuyen en parte esta abundante terminología a la falta de un marco teórico adecuado y consensuado que sirva de referencia a cualquier estudio de restauración. Es necesario llegar a consensos para contribuir a la consolidación de la ecología de restauración como una disciplina científica que vaya más allá de la experimentación *in situ* y de la acumulación empírica de conocimientos y técnicas útiles para cada ecosistema (Hobbs y Norton 1996, Bradshaw 2000).

Existen diversos casos en la literatura en los cuales se enlistan y definen distintas actividades y ejemplos que se pueden tomar dentro del significado de la restauración ecológica (Hobbs y Norton 1996, Vázquez-Yanes *et al.* 1999). En estos trabajos queda claro que el objetivo de la restauración es retornar a las condiciones existentes en las comunidades naturales originales de cada región, incluida la diversidad biológica antes de la degradación. Sin embargo, los autores reconocen que esto no siempre será posible y que en ocasiones será suficiente con recuperar las principales funciones del ecosistema original y algunas veces bastará con desarrollar un paisaje atractivo y salubre para reemplazar otro que no lo es; por ejemplo, un relleno sanitario o una mina.

Hobbs y Norton (1996), reconocen que el término restauración puede ser aplicado a diversas situaciones, encontrándose todas dentro de la única definición avalada por la Sociedad Internacional de Restauración Ecológica; **el proceso de asistir en la recuperación de un ecosistema que ha sido degradado, dañado o destruido** (SER 2004). De esta manera, la restauración persigue siempre alguna de las siguientes metas: restaurar sitios altamente degradados con una ubicación puntual (tal es el caso de las minas), mejorar la calidad de tierras productivas degradadas por agricultura, ganadería o deforestación (tierras erosionadas o salinizadas), incrementar el valor de conservación de zonas protegidas (eliminación de especies invasoras o de algún agente causante de perturbaciones bajas a moderadas), y por último, para incrementar el área de conservación en zonas productivas (reintegrando sitios perturbados a zonas de conservación).

Ante tal cantidad de ejemplos y escenarios donde las actividades de restauración son necesarias y deseables, es que surge una gran cantidad de términos alternativos e intercambiables, algunos equivalentes entre sí, que son utilizados a veces sin consenso, inclusive en sectores especializados como el científico y académico. Expresiones tales como rehabilitación, recuperación (reclamation), regeneración, reasignación, repoblamiento vegetal (revegetación), regeneración, mitigación, remediación, ingeniería ambiental e inclusive actividades como la reforestación, silvicultura, agroforestería y agrosilvicultura pueden traslaparse y, a pesar de que algunas son claramente distinguibles, pueden calificar como restauración de acuerdo a la amplia definición aceptada por la SER. Sin embargo, cada término tiene su propio significado y no es deseable que se usen indistintamente para las mismas situaciones (SER 2004, Monroy Ata 2002, Bradshaw 1984, Bonfil *et al.* 1997).

La Sociedad Internacional de Restauración Ecológica (SER), que fue fundada en 1987, cuenta actualmente con miembros en 37 países y publica la revista especializada en el tema: **Restoration Ecology (**dos revistas especializadas adicionales son **Ecological Restoration** publicada por la Universidad de Wisconsin-Madison y **Ecological Management and Restoration** publicada por la Sociedad Ecológica de Australia), ha definido algunos de los términos mencionados en el párrafo anterior.

De acuerdo a la SER, la restauración busca retornar a las condiciones existentes en las comunidades naturales antes de la degradación, incluyendo su composición, estructura y función. La rehabilitación, al igual que la restauración, se basa en las condiciones pasadas del ecosistema pero ambas difieren en los objetivos y estrategias, ya que la **rehabilitación** se enfoca a la reparación de la productividad, los procesos y servicios de los ecosistemas pero con poca atención a la diversidad biológica original.

En las revistas especializadas, el término rehabilitación ha sido usado principalmente en dos vertientes bastante diferentes: por un lado, para referirse a las actividades realizadas en minas de arena, bauxita, suelos acidificados, suelos con altas cantidades de sodio, sitios contaminados con metales pesados, en resumen; sitios bastante degradados (van Hamburg *et al.* 2004, Jim 2001, De Grant y Loneragan 2003, Weiermans y van Aarde 2003, Redi *et al.* 2005, Tripathi y Singh 2005, Nichols y Nichols 2003, Garg 1999, Bisevac y Majer 1999, Whiting *et al.* 2004, Roche *et al.* 1997, Koch *et al.* 2004). Por el otro, están todas las acciones llevadas a cabo en ecosistemas acuáticos como manglares, esteros, ríos, arroyos y marismas (Van Den Bergh 2005, Day *et al.* 1999, Brooks *et al.* 2002, Parkyn *et al.* 2003), además de algunos ejemplos escasos de bosques degradados en Australia, pastizales empobrecidos (Rokich *et al.* 2000, Muller *et al.* 1998), así como en áreas agrícolas abandonadas (Campana *et al.* 2002).

Un aspecto importante es que al parecer, el término rehabilitación tiene una connotación un poco diferente en revistas relacionadas con la ecología forestal, ya que es utilizado para referirse a acciones de restauración en sitios forestales con menor degradación tales como; la reconstrucción de ensamblajes de especies de bosque mesófilo, a nivel de paisaje para contrarrestar los efectos de la silvicultura y para mejorar la regeneración en bosques afectados por incendios (Equihua y Suarez-Guerrero 2005).

El término **recuperación (reclamation)** comenzó a usarse hace décadas en las minas del Reino Unido y Estados Unidos, pero actualmente tiene una aplicación más amplia que la rehabilitación y entre sus objetivos se encuentran la estabilización de terrenos, el mejoramiento estético y el retorno a lo que dentro del contexto regional sea considerado útil. Un componente frecuente de la recuperación es la revegetación con una o varias especies. En las revistas especializadas el uso dado a este término concuerda con el mencionado anteriormente por ejemplo; suelos salinos después de la extracción de arena (Purdy *et al.* 2005), dunas costeras en Islandia (Greipsson y El-Mayas 2000), sitios con residuos de la extracción de carbón en minas (Hodacova y Prach 2003, Halofsky y McCormick 2005, Williams 2002, Terrance *et al.* 1999), sitios abandonados donde hubo extracción de piedra caliza (Wheater y Cullen 1997, Clemente *et al.* 2004), revegetación con especies riparias en minas de grava abandonadas (Roelle y Gladwin 1999), hidrosembrado en taludes de autopistas (Andrés y Jorba 2000) y con menor frecuencia en la fertilización del suelo y sembrado de especies herbáceas en sitios desprovistos de vegetación (Gretarsdottir *et al.* 2004).

Aparentemente, hay un consenso más o menos claro sobre el significado de algunos de los conceptos mencionados con anterioridad. En la mayoría de los estudios de caso revisados, el término utilizado coincide con aquellos establecidos por la SER. Tal es el caso de la rehabilitación, que aunque ha sido usada tanto para minas como ecosistemas acuáticos, el objetivo principal de la mayoría de los trabajos era restablecer algunas funciones ecosistémicas, además de mejorar la fisonomía y estética de los sitios. Para tal efecto, en los casos de extracción mineral y otros similares, las acciones principales consistieron en el plantado de especies adecuadas para las condiciones del sitio, acompañado de algunas acciones de ingeniería ambiental como movimiento de materiales, rediseño de topografía y remoción de material residual no deseable producto de las actividades extractivas. En los ecosistemas acuáticos sucedió algo similar, ya que la gran mayoría de los estudios partían de sitios mediana a severamente degradados, especialmente en pantanos y ciénegas cercanos ciudades, en los que también era necesario comenzar con acciones de ingeniería para la apertura de canales y restablecer la comunicación y el flujo hidrológico. Sólo algunos estudios se salen claramente del contexto conceptual aquí revisado, como sucedió en algunos trabajos en los que se llama rehabilitación al restablecimiento de ensamblajes de especies en sitios no muy degradados y esto corresponde claramente a un caso de restauración ecológica en su sentido estricto (Equihua y Suarez-Guerrero 2005).

El término restauración se acepta claramente como el que refleja con mayor amplitud la intención de recuperar la integridad de los ecosistemas, ya que abarca los aspectos de composición, estructura y función.

Importancia de un marco conceptual para la Ecología de la Restauración.

En la sección anterior se comentó sobre la necesidad de llegar a consensos para contribuir a la consolidación de la ecología de restauración como una disciplina científica que vaya más allá de la experimentación *in situ* y de la acumulación empírica de conocimientos y técnicas útiles para cada ecosistema (Hobbs y Norton 1996, Bradshaw 2000). Sin embargo, se podría pensar que un marco conceptual no es un requisito indispensable para comenzar a desarrollar una disciplina científica e inclusive, que su repetida demanda reflejaría una actitud más relacionada con la ecología básica actual (Temperton *et al.* 2004). No obstante, es claro que el gran progreso alcanzado en las últimas décadas en el entendimiento de la complejidad de los ecosistemas se dio cuando se pasó del enfoque descriptivo al conceptual. La falta de bases teóricas en la restauración impide la retroalimentación acerca de las experiencias prácticas, limita la posibilidad de verificar la teoría ecológica, aísla a los practicantes de la restauración ecológica y los fuerza a aprender mediante acierto y error y también provoca que toda la información generada no pueda ser transferida para ser usada en otros ecosistemas. Como consecuencia, la ecología de restauración queda resumida a manera de intercambio de historias pequeñas que no contribuyen con mucho a su desarrollo ya que muchos supuestos en proyectos de restauración suelen basarse en conceptos ya rebasados o modificados acerca de cómo funcionan los ecosistemas (Hobbs y Harris 2001).

Conceptos emergentes en la Ecología de Restauración.

A pesar del desarrollo de varios modelos que han intentado organizar la investigación en restauración ecológica haciendo énfasis en el desarrollo de los ecosistemas degradados en presencia y ausencia de restauración, así como los escenarios probables a los que podrían llegar estos sitios (Hobbs y Norton 1996, Bradshaw 1984), predecir los resultados de un esfuerzo de restauración en un sitio en particular es aún difícil (Zedler y Callaway 2000, Young 2005). Lo anterior se puede atribuir a dos causas principales: Primero, a que las condiciones que se encuentran en los sitios de restauración rara vez coinciden con las condiciones descritas por los procesos sucesionales (Zedler 2000) y por lo tanto, la dinámica de muchos procesos ecosistémicos en sitios bajo restauración ecológica es desconocida, además de que la restauración es un proceso más determinístico que la sucesión ecológica. Segundo, las condiciones de muchos sitios en donde se llevan a cabo proyectos de restauración son tan severas, que llevar al sistema a condiciones similares a las históricas o las de sistemas naturales de referencia existentes es imposible (Lindig-Cisneros y Zedler 2000).

La teoría de la sucesión ecológica, y más recientemente los modelos de Estados y Transiciones (state and transition models), han sido incorporados a la base conceptual de la restauración (Hobbs y Norton 1996, Young *et al.* 2005) ya que permiten contar con un marco conceptual para mejorar la predicción de los procesos de restauración. Estos modelos reconocen que existen múltiples estados en un ecosistema y que pueden existir dinámicas no lineales o transiciones (umbrales *sensu* Hobbs y Norton 1996), que pueden dificultar o impedir que se retorne a estados sucesionales avanzados o deseables (ya sea estructural o funcionalmente) desde estados degradados. El concepto de umbrales tiene mucha relevancia para la restauración, ya que explica por qué varias experiencias de restauración no terminaron conforme a los resultados esperados. Estos modelos reconocen que algunos de los estados pueden representar condiciones ajenas a la dinámica natural del ecosistema como consecuencia de la perturbación humana (por ejemplo cuando se introducen especies exóticas invasoras), e incluso que algunos estados pueden ser irreversibles (Zedler 2000). Además, el concepto de umbrales es relevante también porque su presencia podría estar relacionada con la composición funcional del ecosistema, es decir, la transición entre estados puede ser más complicada si involucra cambios de los grupos funcionales representados; por ejemplo, es más difícil sustituir un pastizal por un matorral que un pastizal por otro pastizal (Hobbs y Norton 1996).

Adicionalmente, el reciente desarrollo de la teoría de ensamblaje y la importancia potencial de estados alternativos estables, han facilitado la producción de una gran cantidad de libros y artículos que dan mayor apoyo a la idea que los ecosistemas son dinámicos y pueden tener múltiples trayectorias que conduzcan a numerosos escenarios posibles (Hilderbrand *et al.* 2005). La creencia generalizada de que una comunidad o ecosistema se auto organizará de manera predecible con el paso del tiempo se contrapone con lo observado en numerosos proyectos de restauración ecológica en los cuales se han registrado variaciones no esperadas en la trayectorias de recuperación de dichos ecosistemas. Específicamente en la restauración de humedales se ha seguido la creencia de que basta con restaurar las condiciones físicas (profundidad, calidad del agua y flujos de circulación) para que la parte biótica comience a responder y sin embargo los fracasos o escenarios inesperados han sido numerosos.

Como consecuencia de que las medidas necesarias para transitar de estados degradados hacia estados deseables pueden variar tanto en número como intensidad (Zedler 1999), en función del nivel de perturbación y de la variabilidad temporal de las condiciones bióticas y abióticas, es que el manejo adaptativo de la restauración (ó restauración adaptable) se vuelve la estrategia más eficaz, tanto para generar técnicas como para llegar a las metas deseadas. La restauración adaptable o adaptativa es un esquema de manejo aplicado a la restauración, que consiste en la implementación de una serie de medidas alternativas, la evaluación de sus resultados y la integración del conocimiento adquirido a etapas subsecuentes del manejo del ecosistema para dirigirlo hacia las metas deseadas de restauración (Christensen *et al.* 1996, Zedler 2003). El manejo adaptable en la restauración es un concepto muy importante, porque no debemos olvidar que finalmente estamos extrapolando conceptos muy simplificados y se incrementa la probabilidad de fallas y fracasos. Es por esto que las experiencias de restauración no deben ser por ningún motivo eventos de una sola ocasión, sino que por el contrario, requieren atención periódica y manejo adaptable para incrementar así la oportunidad de diseñar proyectos más exitosos y realistas, ya que suele trabajarse en condiciones de alta incertidumbre debido al poco conocimiento de la respuesta de los ecosistemas a dichas actividades (Hilderbrand *et al.* 2005).

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CAPITULO 2. NATIVE PINE SPECIES PERFORMANCE IN RESPONSE TO AGE AT PLANTING AND MULCHING IN A SITE AFFECTED BY VOLCANIC ASH DEPOSITION

Abstract. Under heavily disturbed conditions, the selection of the appropriate native species and of planting and ameliorating techniques is necessary. Volcanic eruptions create harsh conditions that can preclude native plant establishment. We tested the performance of two native species *Pinus pseudostrobus* and *P. montezumae* for restoration of volcanic ash covered areas. Two age classes of *P. pseudostrobus* and one of *P. montezumae* were tested as well as the effect of mulching to ameliorate harsh substrate conditions. Results show that older plants of *P. pseudostrobus* (19-month old at planting) have higher survival and growth rates than young plants (8-months at planting). Plants at least 19-months-old at planting of *P. pseudostrobus* and *P. montezumae*, are appropriate for restoration of volcanic ash covered areas. Mulching had no effect on plant survival or growth for this experiment.

Keywords restoration, reforestation, soil amendment, native species

Introduction

Forest restoration on severely disturbed soils within fragmented natural areas can be limited by several factors, particularly the nature of the substrate left after the disturbance (Guerrero-Campo and Montserrat-Marti 2004). Under heavily disturbed conditions in highly diverse forests, the selection of appropriate native tree species and planting techniques is necessary. Several factors have been successfully manipulated to aid plant survival and growth, particularly ameliorating harsh substrate conditions by adding soil amendments (Lunt and Hedger 2003, Haas and Kuser 2003) or mulching (*e. g.* Paschke *et al.* 2000, Petersen *et al.* 2004, Blanco-García and Lindig-Cisneros 2005).

Volcanic eruptions create adverse conditions for plant establishment, especially where tephra is deposited (Titus and Tsuyuzaki 2003). The Paricutín volcano, in Michoacán, México, erupted intermittently between 1943 and 1952, substantially disturbing local plant communities and agricultural fields. Studies showed increased species richness and tree canopy cover with time in areas where human disturbance was low or nonexistent, particularly in forested areas (*e. g.* Eggler 1963, Rejmánek *et al.* 1982, Giménez de Ascárate *et al.* 1997). In contrast, agricultural fields and other barren areas that were covered by tephra still had low plant cover (less than 10%) 50 years after the eruption ended, and the vegetation was patchy in distribution and lacking in late-successional species (Lindig-Cisneros *et al.* 2006). In these areas, locally known as “arenales”, several reforestation efforts failed because of low survival and limited growth of surviving plants, except for sites with very shallow tephra, and those close to forest edges (Lindig-Cisneros *et al.* 2002).

The communal lands of the Comunidad Indígena de Nuevo San Juan Parangaricutiro (CINSJP) were affected by the eruption of the Paricutín volcano, and several large arenales persist as is common in volcanic areas of the Americas. The CINSJP practices sustainable forest management for timber extraction and pitch production, as well as ecological restoration for ecotourism. In 2002 we began a restoration experiment in the area, where pine-bark mulch increased survival and growth of *Pinus pseudostrobus* Lind. After the first year of planting, the survival rate of mulched trees doubled the survival rate of non-mulched trees (Blanco-García and Lindig-Cisneros 2005). Based on this study, a second experiment, reported here, tested the effect of pine-bark mulching on survival and growth of the natives *P. pseudostrobus* and *P. montezumae* Lamb. for four successive growth seasons.. These pine species are native and abundant in central México. The objectives of this study were to assess the performance of these two pine species, the age at planting, and efficacy of mulching as a soil amendment. We hypothesized that mulching will have a differential effect on survival and growth of the pine species tested because of the different growth habits*. Pinus pseudostrobus* is a fast growing species (López-Upton 2002), and *P. montezumae* is a slow growing species during early development. Young *P. montezumae* occurs in a "grass state" that can persist for several years because there is a below-ground investment in a tap root. Presence of a grass state in *P. montezumae* has been suggested as an adaptation to chronic fire (Rodríguez-Trejo and Fulé 2003), similar to longleaf pine, *Pinus palustris* Mill. (Ramsey *et al.* 2003).

Methods

The study area is located in the Comunidad Indígena de Nuevo San Juan Parangaricutiro (19° 30´ N, 102° 12´ W; 1900 to 3200 m asl) in the State of Michoacán, México. Pine and oak-pine forests dominate the area, mean annual temperature is 15.1°C and mean annual precipitation is 1460 mm; the dry season extends from November to May, less than 10% of the precipitation occurs during winter. As a consequence of the Paricutín volcano eruption, large areas covered with volcanic ash of variable depth exist within the Nuevo San Juan forests. Several such areas, the “arenales”, consist of bare sites (mostly agricultural fields at the time of the eruption) covered by a layer of black-colored volcanic ash of basaltic origin where sparse vegetation has established itself, resulting in isolated patches surrounded by large expanses of fine grained material. All vegetation patches are dominated by shrubs, mostly *Eupatorium glabratum* Kunth, and only one species of a leguminous shrub is present, *Lupinus elegans* Kunth. No tree species have established after 50 years following the end of the Paricutín volcano eruption (Lindig-Cisneros *et al.* 2006).

The experiment consisted of 72 square plots of 1.96 m^2 within a 0.5 ha fenced area to exclude cattle within the volcanic ash deposit locally known as “Mesa de Cutzato” (19° 30´ 42.4” N, 102° 12´ 03.0” W; 2450 meters in elevation). Plots were interspaced 1 m apart from each other forming a regular grid. We designed a two-factor experiment to test performance of three stock types of two native pine species, *P. montezumae* 19-months old at planting (“large” *P. montezumae*), and *Pinus pseudostrobus* of two ages at planting, 8 and 19 months-old (“small” and “large” *P. pseudostrobus*) as the first factor; and presence-absence of pine-bark mulch as the second factor. Two different ages at planting of *P. pseudostrobus* were tested because local nurseries produce 8 and 19 month-old stock for this species. Plots in the field were assigned randomly to each of these treatment combinations. Two pines of the same species-age-class were planted per plot (at a distance of 1 m). Pines were planted and mulch was added during the third week of June, 2003. A total of 144 plants were obtained from the nursery of the CINSJP**.** Pine seeds for propagation at the nursery were collected from nearby forests. Plants were produced in nursery bags, 8-month *P. pseudostrobus* in 7 X 15 cm plastic bags (230 cm^3) and 19-month plants of both species in 10 X 15 cm plastic bags (470 cm^3); larger bags are used for the latter because they have to be kept in the nursery for a longer time than 8-month plants and small containers prevent good root development. Mulch consisted of pine bark from the CINSJP saw mill; size ranged from 0.5 cm to 4 cm in diameter. A layer of 2-3 cm was applied to mulched plots.

Pine plants were evaluated on five occasions: first in July 2003 for initial height, and afterwards once a year during the month of October, from 2004 to 2007. Air and volcanic ash temperatures (four centimeters below ground surface) were recorded hourly with data loggers programmed to do so (Hobo® H01-001-01 Onset Computer Corporation, USA) over four years, because our previous study showed that these variables are correlated with pine mortality (Blanco-Garcia and Lindig-Cisneros 2005).

For statistical analyses, because each experimental unit consisted of a plot with two pines of the same species and age class in the case of *P. pseudostrobus*, pine mortality was analyzed using generalized linear models (GLM) for Poisson distributed data with three states possible for each plot: 0, 1 or 2 dead pines (Dobson 2002).

Height for plants 19 months old at planting was analyzed with ANCOVA for the data of each year using as covariable the height that plants had when the experiment was set up (height at planting), and as explanatory variables species and mulching.

When both pines in a plot survived, the mean height was used as the datum for the plot. Because mulching was not significant in all analyses, a simplified model with the covariable and species is reported. Repeated measures ANOVA was not applied because the data did not comply with most of the assumptions for this analysis, particularly with the assumption of circularity (Crowder and Hand 1990). Residuals were checked for compliance with ANOVA assumptions. All analyses were carried out using R (R Development Core Team, 2007).

Results

In our experimental site, lower substrate temperatures were recorded in April in 2005 when compared to those recorded in 2004. For 8 days in April 2004, substrate temperatures reached or exceeded 50^{o}C, and during April 2005, substrate temperatures never reached 50^{o}C.

Mortality after the first growing season (October 2004) differed between pine species and between the age classes of *P. pseudostrobus* (Table 1). Only 4 % of *P. montezumae* plants in non-mulched plots died, and none in mulched plots. After the same growing season, 4% of the large *P. pseudostrobus* plants died, both in mulched and non-mulched plots. As regards small *P. pseudostrobus* plants, 17% died in mulched plots and 21% died in non-mulched plots. The analysis of deviance shows that mortality differences among species age-classes were significant (Table 2a) but not between mulched and non-mulched plots. After the second growing season, mortality for both species increased, also increasing for both age classes of *P. pseudostrobus. Pinus montezumae* mortality increased to 21% in non-mulched plots and to 17% in mulched plots. For large *P. pseudostrobus* plants, mortality reached 21% in non-mulched plots and to 12% in mulched plots. The highest mortality was for small *P. pseudostrobus* plants, reaching 92% in both mulched and non-mulched plots. For the second growing season, mortality by species was statistically significant, but not among mulched treatments (Table 2b). After the third growing season, all small *P. pseudostrobus* plants were dead except for one. There was no additional mortality for large *P. pseudostrobus* plants, and *P. montezumae* mortality increased to 29% only in non-mulched plots. No significant differences between species or mulching were found for large plants of both species at planting after the third growing season. No further mortality occurred after the last growing season of the experiment.

Height differences among planted stock occurred for the first two growing seasons, when comparing large plants of *P. pseudostrobus* and *P. montezumae* as shown by analysis of covariance (Table 3). In 2004, after the first growing season, *P. pseudostrobus* plants were on average 64 ± 2 cm tall, and *P. montezumae* 28 ± 2 cm tall. After two growing seasons, in 2005, *P. pseudostrobus* plants were on average 114 ± 7 cm tall, and *P. montezumae* 59 ± 5 cm tall. After the third and fourth growing seasons (2006 and 2007) *P. pseudostrobus* plants were also taller than *P. montezumae* plants (211 ± 14 cm *vs.* 126 ± 9 cm, and 290 ± 17 cm *vs.* 198 ± 14 cm), but because of the large variances in height for both species the differences are not significant.

Discussion

Plant survival differed between species and between *Pinus pseudostrobus* age classes. Small *P. pseudostrobus* plants were not appropriate for restoration of volcanic ash covered areas because of their low survival rate. The use of mulching did not increase survival for the species planted; this contrasts with a previous experiment in the same "arenal" (Blanco-García and Lindig-Cisneros 2005), when pine bark mulching significantly increased plant survival. The difference in the magnitude of the effect of mulching may be a consequence of inter-annual variation in high substrate temperatures during the peak of the dry season in April. The year when the experiment reported here was planted (2003), corresponding to the first growing season of the previously published experiment, was a particularly hot year. During April 2003, 50^{o}C of temperature or more was reached 19 times, when measured 4 cm below the surface of the volcanic ash, causing high plant mortality in non-mulched plots of the previously published experiment. In contrast, the first growing season (2004) that the plants of the current experiment experienced was comparatively milder, with only eight days with high temperatures at or higher than 50^{o}C. Furthermore, by the second growing season (2005), not a single time was 50^{o}C reached. Annual precipitation did not significantly differ in our study site from 2003 to 2006. The month of April corresponds to the peak of the dry season since rains can start as early as June. Consequently, soil moisture was very limited in the course of this month during all the growing season. Our previous experiment in the same site, that was monitored every two months, showed that pine mortality peaks in April, suggesting direct damage of high substrate temperatures to stem tissues.

During the length of the experiment, large (19-month-old plants at planting) of *P. montezumae* were shorter than large plants of *P. pseudostrobus.* Height significantly differed between species for the first two growing seasons, because *P. montezumae* growth was very limited during this period. Studies for other pine species that present a grass state have shown that low nutrient availability (Koskela 2000, Prior *et al.* 1997), competition (Nelson *et al.* 1985, Ramsey *et al.* 2003), and site preparation (Knapp *et al.* 2006) can affect emergence from the grass state. At our site the harsh conditions of the sandy volcanic ash substrate might be responsible for the prolonged grass stage of *P. montezumae* plants that persisted for two growing seasons after planting at the field site, and more than three and a half years after the seeds emerged in the nursery. At less demanding sites in the region, *P. montezumae* overcomes the grass state after one growing season in the field (Viveros-Viveros *et al.* 2007). Our results suggest that both *P. pseudostrobus* and *P. montezumae* are appropriate for restoration of volcanic ash covered areas as long as plants are at least 19-months old when initially planted. *Pinus pseudostrobus* that are 8 months old at planting are not appropriate because they are either too small to withstand the demanding environmental conditions of the “arenales” or because the low volume of substrate in which they are propagated provides fewer resources in the field than the volume of substrate that 19 month old plants have.

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Table 1. Plant mortality and mean plant height (± standard error) of surviving plants after four growing seasons. Age class corresponds to age of plants at planting.

		Mortality (%)			Height (cm)		
Year	Mulching	*P. montezumae*	*P. pseudostrobus*		*P. montezumae*	*P. pseudostrobus*	
		19 months	8 months	19 months	19 months	8 months	19 months
2004	Present	0	17	4	28±3	19±1	63±3
	Absent	4	21	4	28±4	18±1	64±3
2005	Present	17	92	12	60±6	25±5	107±9
	Absent	21	92	21	58±9	25±2	121±11
2006	Present	17	96	12	128±13	-	204±21
	Absent	29	100	21	123±14	-	219±20
2007	Present	17	100	12	210 ± 20	-	280 ± 26
	Absent	29	100	21	186 ± 20	-	300 ± 23

Table 2. Analysis of Deviance table for mortality after (a) the first (2004) and (b) the second (2005) growing seasons.

	D.f.	Deviance	Residual D.f.	Residual Deviance	P (χ^2)
a: 2004 mortality					
Null			71	48.55	
Species-age-class	2	9.05	69	39.49	0.01
Mulching	1	0.33	68	39.16	0.56
Interaction	2	1.16	66	37.99	0.56
b: 2005 mortality					
Null			71	86.77	
Species-age-class	2	38.33	69	48.43	<0.01
Mulching	1	0.02	68	48.42	0.90
Interaction	2	0.60	66	47.81	0.74

Table 3. ANCOVA table for height after the first growing season for *P. pseudostrobus* and *P. montezumae* both 19 month old at planting. Height at planting was used as covariable and data were square-root transformed.

	D.f.	Deviance	Residual D.f.	Residual Deviance	F	P
Height at 2004						
Null			40	97.3		
Height at planting	1	81.3	39	16.0	248.0	<0.001
Species	1	3.6	38	12.4	10.9	0.002
Height at 2005						
Null			40	182.6		
Height at planting	1	87.9	39	94.7	39.1	<0.001
Species	1	9.3	38	85.3	4.2	0.048
Height at 2006						
Null			40	297.0		
Height at planting	1	94.2	39	202.8	19.4	<0.001
Species	1	18.3	38	184.4	3.8	0.059
Height at 2007						
Null			40	312.3		
Height at planting	1	73.8	39	238.5	12.6	0.001
Species	1	15.8	38	222.7	2.7	0.109

CAPITULO 3. RESTORATION OF FOREST COVER UNDER STRESSFUL ENVIRONMENTAL CONDITIONS: RESISTANCE TO FROST DAMAGE OF THREE CONIFER SPECIES.

Abstract. Survival, growth and resistance to frost damage of two native pine species (*Pinus pseudostrobus* Lind. of two ages at planting, and *P. montezumae* A. B. Lambert) and one non-native pine species (*Pinus greggii* var. *australis* Donahue et Lopez) were quantified in an ecological rehabilitation site prone to stressful environmental conditions (frequent freezing temperatures) and with possible water stress (due to a teprha layer of volcanic origin) .

The site was followed for a three year period. Restoration treatments were as follows: twelve reforested plots with an induced lupine shrub cover (*Lupinus elegans* Kunth), twelve plots with pine-bark mulching and 12 plots as control.

After three years, we found significant advantages of planting *P. montezumae* on sites with frequent freezing temperatures with regard to other pine species. We did not corroborate the negative relationship between frost resistance and growth rates recorded on other studies for this species. We also found higher growth rates in *P. montezumae* than in *P. greggii* and *P. pseudostrobus.* No advantages of using a bigger plant size of *P. pseudostrobus* were found in this site. Instead we found higher frost damage in *P. pseudostrobus* of 17 months-old at planting than in plants 7 months at planting.

The harsh environmental conditions did not allow us to test the effects of a nurse-plant interaction in conifer plants (freezing temperatures killed *Lupinus elegans*)

neither the use of a mulching to prevent drought stress of conifers (heavy run-off carried out the pine bark mulching).

Keywords: ecological restoration, Nuevo San Juan Parangaricutiro, frost damage, tephra deposits, *P. montezumae, P. pseudostrobus* and *P. greggii*.

Introduction

Frost damage is one of the causes of seedling mortality in reforestations in Mexico at high altitudes (Sáenz-Romero *et al.*, 2003, Sáenz-Romero *et al.,* 2007, Sáenz-Romero *et al*., 2008). Average pine seedling mortality, one year after establishing the plantation, is 62% in the State of Michoacán, Western Mexico: Frost damage causes 14% of it. Furthermore, the combined effect of inadequate species selection, drought stress and frost damage causes 46% mortality (Sáenz-Romero and Lindig-Cisneros, 2004). Frost damage also causes growth reduction, loss of stem straightness, and increases susceptibility to fungi and other pathogen infections (Alden and Hermann, 1971; Anekonda and Adams, 2000). Therefore, the selection of appropriate species adapted to frost occurrence is a key factor to increase seedling survival and growth in reforestation programs that aim at restoring forest cover. However, species selection requires finding an appropriate balance between growth potential and frost resistance (low damage in plants by freezing temperatures), because it has been shown that species with greater frost resistance also have less growth potential (Rehfeldt, 1983, 1985; Jonsson *et al.*, 1986).

Plant establishment in degraded lands depends on the capability of the selected species to overcome harsh conditions. When conditions are extreme, some responses, like growth, may be inhibited even for tolerant species because of the high stress levels in response to the physical environment. Cover and mulching treatments are widely used to improve physical soil conditions, by preventing soil drying and by reducing high soil temperatures (Barradas 2000). Pine bark mulch was selected because it has positive effects on pine survival under harsh conditions (Blanco-García and Lindig-Cisneros 2005) and because it's a leftover product of local timber industry.

As part of a participative research effort using an adaptive restoration scheme conducted with the Comunidad Indígena de Nuevo San Juan Parangaricutiro (CINSJP) in the state of Michoacán, Mexico, a series of restoration experiments have been implemented since 2001. The CINSJP applies an internationally recognized sustainable forestry model that aims at producing timber and preserving biodiversity (Velázquez *et al.* 2003, Blanco-García y Lindig-Cisneros 2005).

The main goal of the collaborative effort is to transform current reforestation strategies into restoration efforts by adding actions that will lead to more diverse and structurally complex plant communities, but still economically productive, and that will foster establishment of tree species resistant to difficult environmental conditions (freezing temperatures in this case, which limits the current reforestation efforts).

We planned a reforestation using two native and one non-native conifer species (*Pinus pseudostrobus, P. montezumae* and *P. greggii,* respectively). The conifer species were selected for the following reasons; *Pinus pseudostrobus* Lindl., and *Pinus montezumae* Lamb. are two ecologically and economically very important species in the Neovolcanic Axis of central México, due to their relatively large distribution and extended use for saw timber, cellulose and fire-wood (Perry,1991). *Pinus montezumae* and *P. pseudostrobus*, besides being the most common species in the region, are appreciated for timber production. *Pinus greggii* is a native species in the Mexican East Mountain Chain and it was selected to evaluate the performance of a non-native conifer species of wide ecological plasticity and fast growth rates in a highly degraded environment, and also to fulfill the need of local land owners of testing this species.

METHODS

Study Area:

The study area is located in the CINSJP in the Northeast region of the State of Michoacán, Mexico at 2250 m above sea level. Pine-oak forests dominate the area. The community forests (11,694 ha) are managed for timber extraction under sustainable forestry practice. Within the Nuevo San Juan forests and in neighboring communities, large areas covered with volcanic sand (tephra) deposits of variable depth exists, as a consequence of the Paricutín volcano eruption and are locally known as “arenales”. The arenales consist of bare areas (mostly agricultural fields at the time of the eruption) covered by a layer of black-colored tephra of basaltic origin where sparse vegetation has established itself, creating isolated patches surrounded by large expanses of the fine grained material. All vegetation patches are dominated by shrubs, mostly *Eupatorium glabratum* H. B. K. Our study site is a plain terrain surrounded by hills and it was covered with 1-1.5 meter tephra layer from de Paricutín Volcano in 1943-1952. Eight years ago the material was extracted and used in construction work, leaving the original soil almost exposed only with a 5-10 cm tephra layer.

Restoration experiment establishment

In June 2004, three coniferous species (*Pinus pseudostrobus, Pinus montezumae* and *Pinus greggii*) were used to set-up a restoration experiment in a 5,000 m^2 area (figure 1). Within the area, thirty-six 8 × 8 m plots were established with a distance of 4 m between adjacent plots, and were divided in two blocks for accounting for differences in distance from forest border and the presence of remnant vegetation left after the removal of the volcanic-sand layer. Four conifer plants of each species: *Pinus greggii* (17 months old), *Pinus montezumae* (19 months old) and *Pinus pseudostrobus* (the latter with two ages at planting; 7 and 17 months-old) were planted inside each plot following a Latin Square design (for a total of 16 plants per plot) with a 2 m distance between adjacent trees. In order to improve microclimatic conditions around plants, six plots in each block were selected at random and uniformly sown with 1,200 seeds of *L. elegans* (a native shrub common in the study area) in order to form a canopy of this species with an average of 1 plant per square meter (previous greenhouse experiments showed that 20 *L. elegans* seeds were needed to one seedling reach the adult size. The area of each plot, $64m^2$, is close to the average size of natural *L. elegans* clumps within the study area. Six additional randomly-selected plots per block were used to test the effect of pine bark as mulching. Mulching was applied by adding a 4-cm deep layer covering 1 m^2 around each tree in the plot. The remaining plots per block were kept as a control. Because all lupines died during the first year, in July 2006 we planted 320 *L. elegans* seedlings (two seedlings around each surviving conifer in legume plots).

Conifer survival and growth were evaluated every two months during three years and four months (from June 2004 until October 2007). Soil and air temperature were recorded hourly with a data logger (Hobo® H01-001-01 Onset Computer Corporation, USA) located four centimeters below soil surface and 1.5 m above the soil respectively. Relative growth rates were calculated using the formula: (RGR = log (h_2)-log (h_1))/ (t_2-t_1), where h_2 is the height at the end of the experiment, h_1 was initial height and t is the time interval in years.

Frost damage, evaluated at 15 months (March 2003), was assessed visually with a 0–10 index, according to the percentage of frost damage in the plant, where 10% was equivalent to 1 and so on, up to a damage of 100%, equivalent to 10. To assess the damage value, each seedling was divided in ten equivalent parts along its vertical axis, from the seedling collar to the tip of the leader bud, then, each tenth part of the seedling was scored for foliage (intense green needles = no damage = 0, brownish or light green needles and/or dehydrated appearance = severe damage = 1, intermediate aspect = 0.5) or for shoot (healthy light brown turgent shoot = no damage = 0, dark brown and/or dehydrated aspect and/or twisted shoot = severe damage = 1, intermediate aspect = 0.5); finally, scores of each seedling part was added up to obtain the frost damage index (method based on Glerum, 1985, en Saénz *et al.,* 2007).

Statistical analysis

Final conifer survival (October 2007) was analyzed using GLM for Poisson distributed data with five possible states for each plot; 0, 1, 2, 3 and 4 live plants (Dobson 2002); and two explanatory variables were explored: treatment, block and their interaction. Conifer survival through the length of the experiment was analyzed with the Kaplan-Meier estimator of survival, a function that describes the distribution of the times of survival, to further explore how the time of survival differed between legume, pine-bark mulching and control plots. Relative growth rates and height gained during the experiment were analyzed with ANOVA using as the response variable the mean value for each plot. When only one tree survived, the datum was used for the analysis; when no trees survived, RGR and height were considered to be zero. All analyses were carried out using S-Plus 2000 (Statistical Sciences 1999).

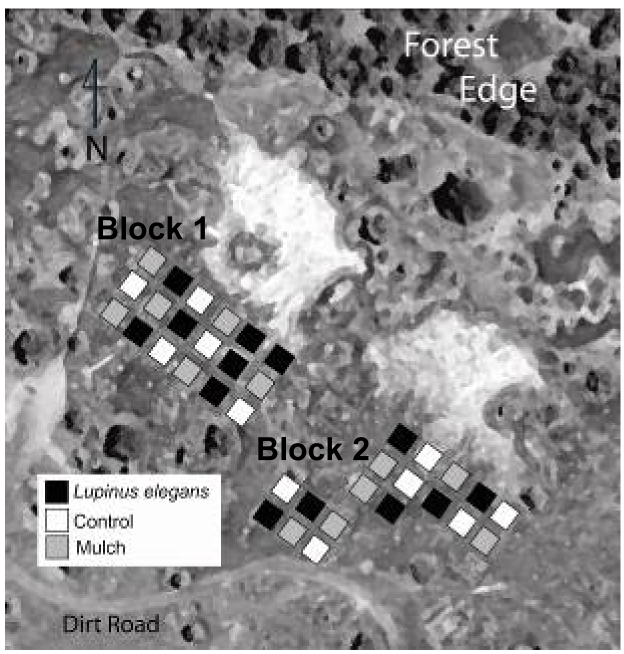

Figure 1. Experimental design of restoration treatments in the study site

RESULTS

Because of the topography of the area (an small valley surrounded by hills), freezing temperatures occurred in consecutive days as a consequence of a thermal inversion occurred during the winter months of every year (Figure 2) and directly affected survival and growth of the conifer species and were responsible of unsuccessful establishment of *L. elegans*. According to air temperature data recorded from November 2005 to January 2006, below zero temperature occurred during 25 days of a total of 60 days during the coldest months in winter (Figure 2). As mentioned above, *L. elegans* canopies did not develop because freezing temperatures during winter prevented this species to reach adult size and died prematurely. Also, pine-bark mulching did not have any effect because of heavy run-off during the rainy season that washed away the material. Both treatments (mulch and *L. elegans*) were present only during six months after the experiment setting (16% of the study time).

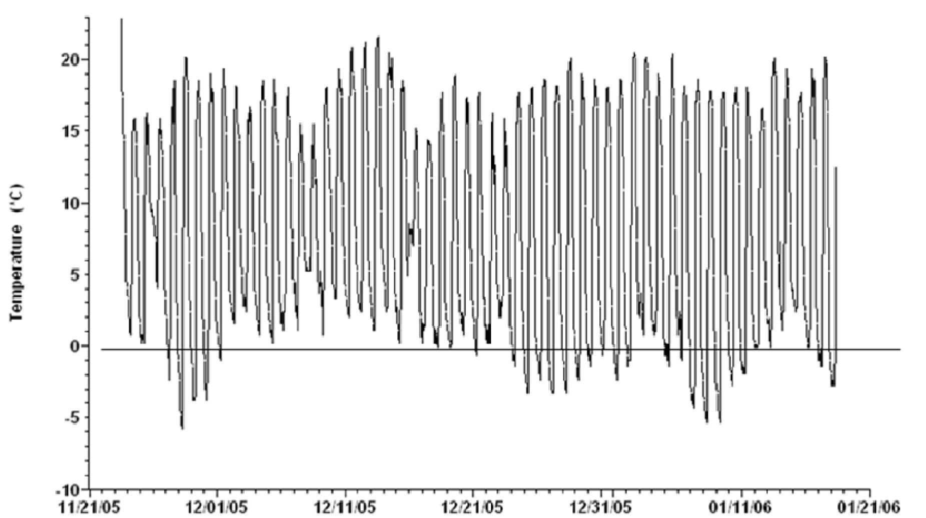

Figure 2. Air temperature during the coldest months of winter 2005-2006

Conifer survival

Overall survival of conifer plants in the restoration experiment was 73%, and significant differences between species performance were found. No effects of treatments or blocks in conifer survival were detected (table 1, figure 3). After three years, *P. greggi* and *P. montezumae* recorded higher survival percentages than *P. pseudostrobus* regardless of the age at planting (96 and 87% against 55% of both ages of *P. pseudostrobus*), (Table 2). According to the Kaplan-Meier estimator, no differences among treatments was found in shape of the survival curves of all conifer species during 2004-2007 but a block effect was detected for *P. pseudostrobus* 7-months-old at planting (60 vs. 32% final survival in both blocks), (table 3).

Frost damage in the conifer species was an important factor affecting plant performance during the whole study. Significant effects of the blocks and species were detected. Damage was more intense during 2005-2006 winter when compared with 2006-2007 winter (no block effect was detected in the latter winter, only species effect), (Table 4).

Pinus pseudostrobus plants were strongly affected by frost damage and apparently it was the main reason of mortality for this species.

Table 1. Probability values of the analysis of deviance applied to the number of survivors conifers per plot.

	p
Block	0.224
Species	0.001
Treatment	0.523
Block- species	0.325
Block- treatment	0.562
Species- treatment	0.966
Block- species- treatment	0.982

Table 2. Percent of survival of conifer species after more than three years (December 2007)

Species	Survival (%)
P. greggii	96
P. montezumae	87
P. pseudostrobus 7 months old	55
P. pseudostrobus 17 months old	55
Average survival	73

Table 3. P values of the Kaplan-Meier estimator of survival curves of conifer species among treatments and blocks during the three years of the study.

Species	Block	Treatment
Pinus pseudostrobus 7 months	0.008	0.606
P. pseudostrobus 17 months	0.528	0.230
P. montezumae	0.999	0.164
P. greggii	0.647	0.812

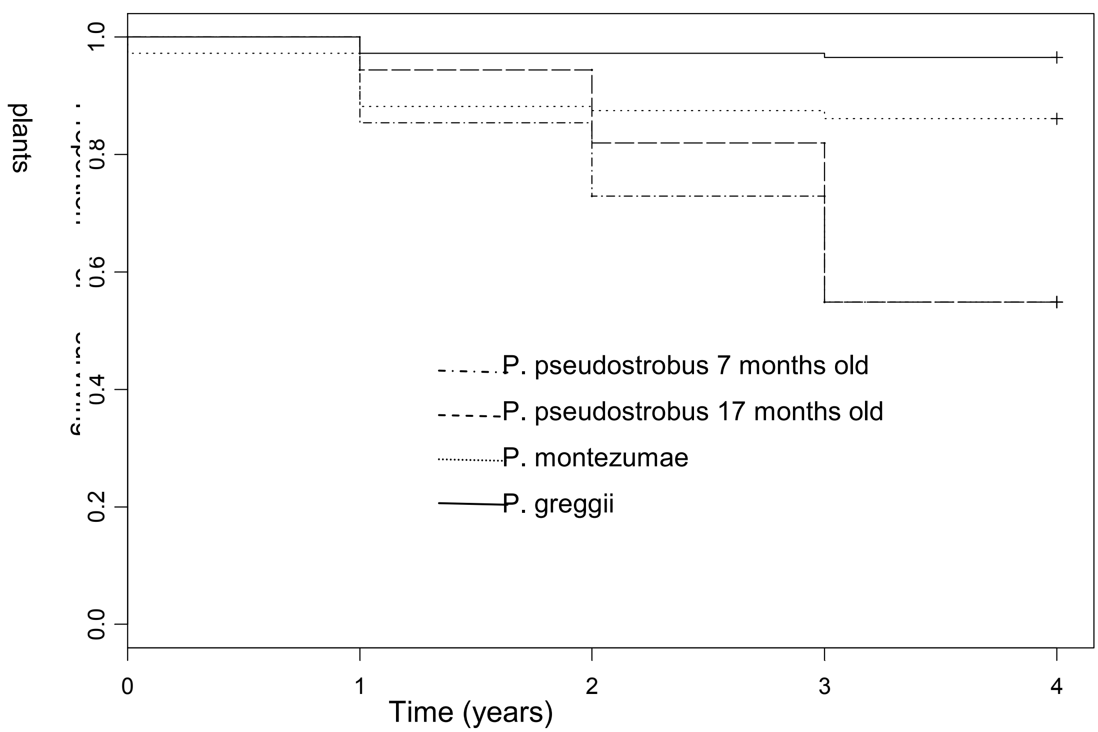

Figure 3. Survival curves of conifer species during the three years of the study.

Table 4. Summary of analysis of variance with frost damage data from two consecutive winters.

Winter 05-06	Df	Sum of Sq	F	P
Block	1	60.7	13.02	0.000
Species	3	2241.1	160.3	0.000
Block: species	3	28.5	2.04	0.106
Residuals	568	2646.9		
Winter 06-07				
Block	1	4.58	8.65	0.003

Conifer growth

A significant species effect was detected on relative growth rates (RGR) and a block: species interaction was detected for *P. montezumae* (Table 5 and 6) which grew shorter in block 2, the more exposed block and also the farther from the forest edge. Although *Pinus greggii* was the tallest species in the experiment (Table 7), it showed prostrated and deformed trunks. Although *P. montezumae* showed very short initial height, due to the "grass stage" (this species frequently shows a physiognomy similar to a grass during the first year of planting), it had the highest relative growth rates after three years (Figure 4).

The magnitude of frost damage in the conifer species was closely related with species identity, block and plant size (Table 4). All the species differed significantly in frost damage; *P. montezumae* was the less affected, and *P. greggii* showed intermediate frost damage, while both *P. pseudostrobus* ages were strongly affected by frost damage (4.6 y 6.4 average frost damage value respectively), the taller plants of *P. pseudostrobus* (17 months old) suffered higher frost damage (Figure 5).

It’s also important to mention that in 2007-2008 a less severe winter occurred and no frost damage was registered in the plants.

Table 5. Summary of a three way ANOVA with accumulated relative growth rates during 2004-2007.

	df	Sum of sq	F	P
Block	1	0.000	0.00	0.929
Species	3	1.022	63.18	**0.001
Treatment	2	0.009	0.84	0.432
Block: species	3	0.044	2.73	*0.046
Block :Treatment	2	0.005	0.48	0.618
Species :Treatment	6	0.003	0.10	0.995
Block:species :Treatment	6	0.008	0.27	0.947
Residuals	120	0.643		

Table 6. Summary of one way ANOVA applied to conifers relative growth rates among blocks.

	F	*p*
P. greggii	0.72	0.40
P. montezumae	8.21	**0.01
P. pseudostrobus 7 months	1.28	0.26
P. pseudostrobus 17 months	0.042	0.83

Table 7. Average height ± se (cm) of conifer species in 2004 and 2007

July 2004	P. greggii	P. montezumae	P. pseudostrobus 7 months old	P. pseudostrobus 17 months old
Block 1	48 ± 1.3	10 ± 0.9	7 ± 0.4	27 ± 0.9
Block 2	48 ± 1.5	10 ± 1	6 ± 0.3	27 ± 0.9
October 2007				
Block 1	176 ± 11.1	120 ± 6.0	26 ± 4.8	55 ± 6.1
Block 2	189 ± 10.4	99 ± 7.0	26 ± 4.2	72 ± 8.8

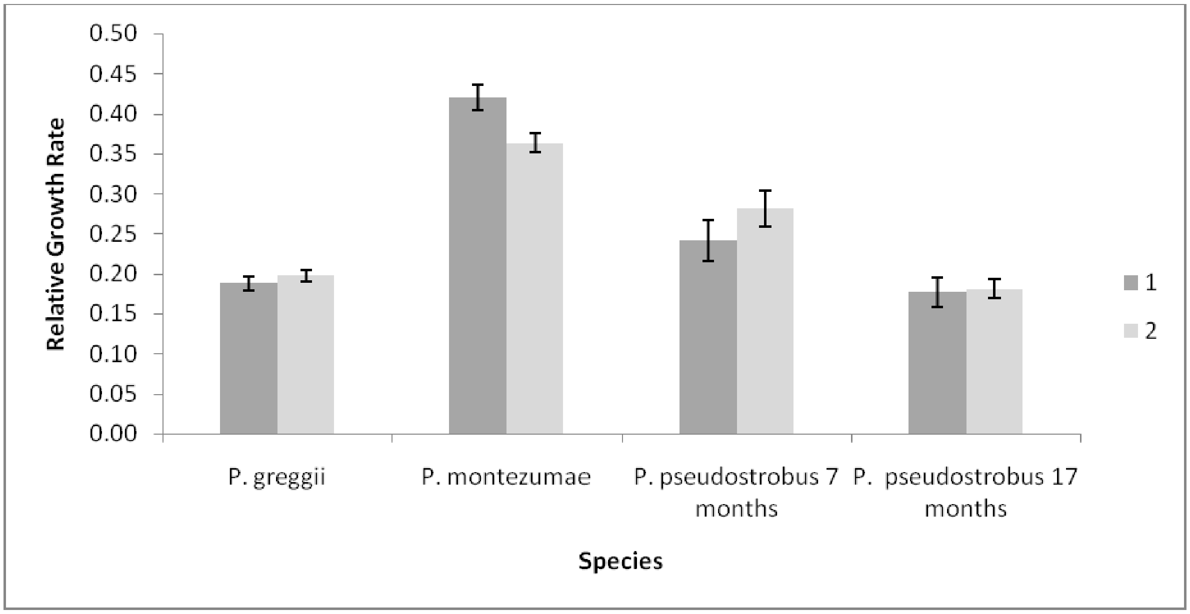

Figure 4. Relative Growth rates of conifer species among blocks (1 and 2) after three years

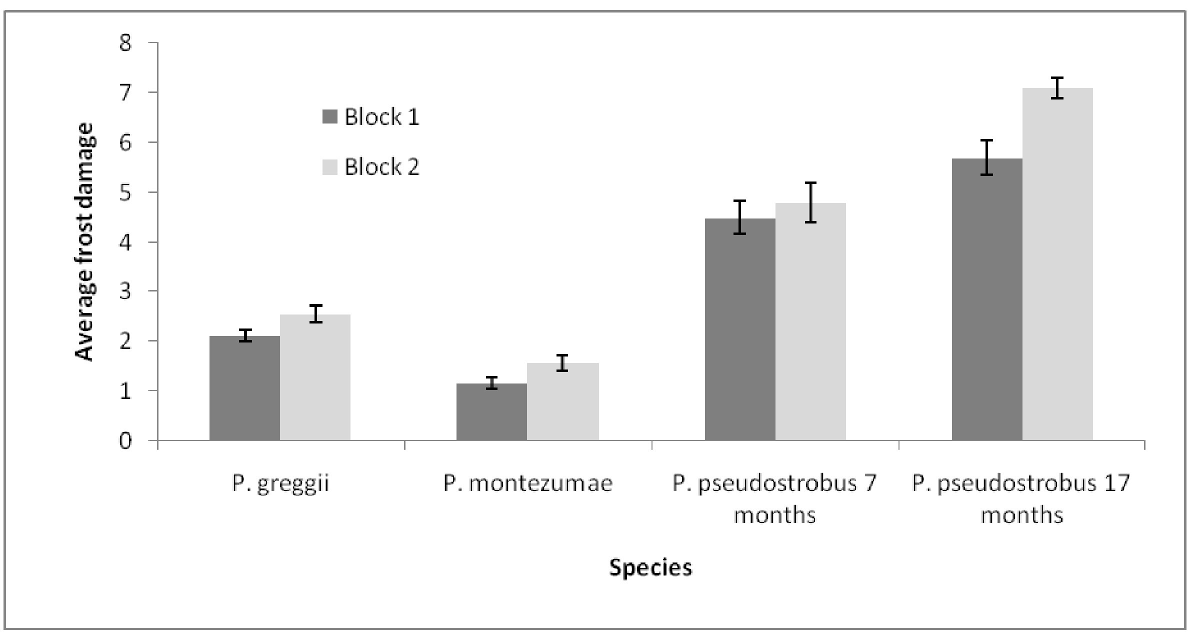

Figure 5. Average frost damage in winter 2005-2006 for species and blocks.

Freezing temperatures registered during winter 2005-2006 were less severe than freezing temperatures of winter 2006-2007, and the height of conifer species was important to frost damage. Even though height of plant at the moment of frozen was significant for frost damage in 2005-2006, it explained weakly this relationship (figure 6). On the other hand, plant height was important in winter 2006-2007 for the occurrence of frost damage (figure 7), especially for *P. greggii* y *P. montezumae* (shorter plants suffered a more severe frost damage than taller plants of both species) and height explained partially the variability in frost damage . Regarding *P. pseudostrobus*, both ages suffered severe frost damage and is not related with the height of plant (figure 7).

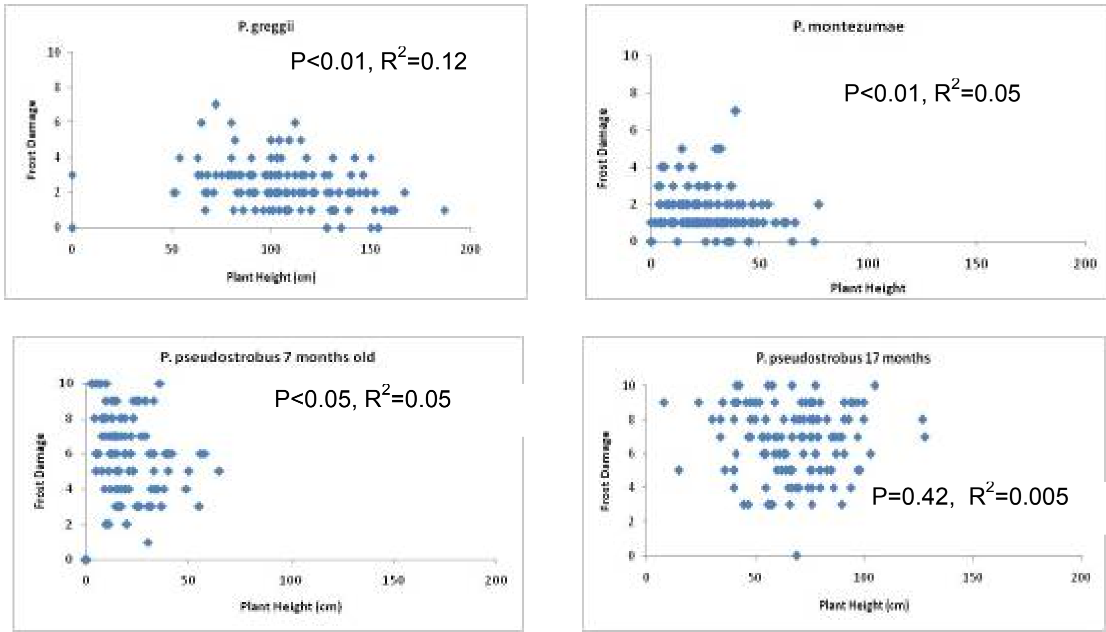

Figure 6. Relationship between frost damage and conifer height in winter of 2006

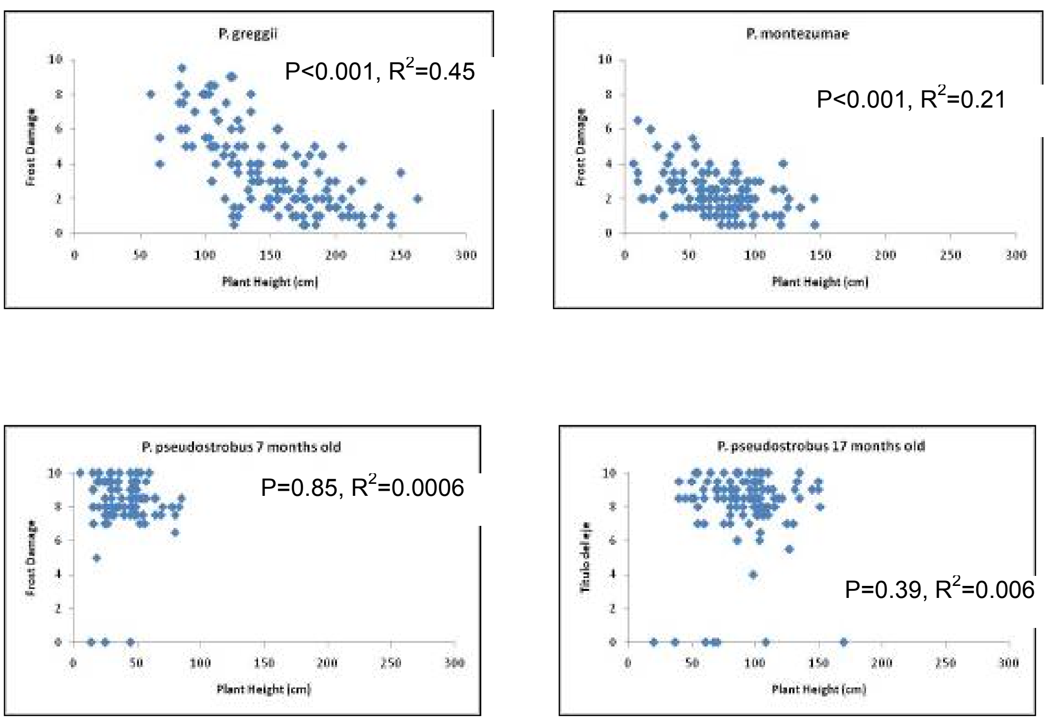

Figure 7. Relationship between frost damage and conifer height in winter of 2007

DISCUSSION AND CONCLUSIONS

Pinus pseudostrobus is the most abundant and economically important pine species in the CINSJP. Usually older plants show better survival and growth when compared with younger plants, because a taller plant can perform better under stressful field conditions. Nevertheless, in our study site, taller plants of this species were more affected by severe frost damage in Winter 2005-2006.

Frost occurrence might be explained by the fact that the site is a flat valley, at a foothill, and during the night the air cools off on the hill and then goes down and settles down at the valley; this type of cold air usually is the cause of frost damage (Viveros-Viveros *et al.*, 2007).

Our results indicate that *P. montezumae* can perform with excellent results because it was the less affected species by frost damage, which is a recurrent condition in the site because of its particular location in the bottom of a small valley.

Compared to the other species used in this study, *P. montezumae* showed larger resistance to freezing temperatures and also had the highest growth rates. Our results are in agreement with Viveros-Viveros *et al.* (2007) who observed negative relationship between frost resistance and growth rates; they reported the higher resistance of *P. montezumae* to freezing temperatures against *P. pseudostrobus* but *P. hartwegii* showed no frost damage (is the pine that grows in México at the highest altitudes; 3000-3660 masl) but at the cost of having smaller growth rates.

With regard to the other species, *P. greggii* showed the highest total height after three years but not in relative growth rate (*P. montezumae* registered a higher RGR) since the beginning of the study, but was more susceptible to frost damage and also showed prostrated trunks following the direction of the dominant winds. Prostration might be a consequence of poor root system development. These results suggest that tephra deposits are not suitable for *P. greggii* establishment.

The observed higher growth rates of *P. montezumae* on our site may be related the initial “grass stage” characteristic of this species. *Pinus montezumae* shows a grass stage during its early years of growth and it has been argued that it is an adaptation both to occurrence of frosts and to wild fires. During the grass stage the root system develops and a reserve builds up that the plant can use for fast growth once the grass stage is broken (Perry, 1991).

For *P, greggii* y *P. montezumae*, the negative relation observed between frost damage and plant height (taller plants with less frost damage) might be explained by the fact that taller plants present a bigger proportion of their canopy far from soil, where low temperature is more severe.

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CAPITULO 4. RESTORATION OF FOREST COVER IN AN AGRICULTURAL ABANDONED FIELD THROUGH A NURSE-PLANT INTERACTION.

ABSTRACT

Nurse-plant effects have been shown to be determinant for successful plant establishment mostly in arid and Mediterranean environments, and also as an effective restoration strategy for sites were the microclimate limits native species growth and survival.

Survival and growth of three different conifer species: *Pinus pseudostrobus* Lind. of two ages at planting*, P. montezumae* A. B. Lambert and *Abies religiosa* **(Kunth) Schltdl. & Cham.** were quantified in an ecological restoration site for assessing the effect of *Lupinus elegans* Kunth as a nurse plant.

The site was followed for a three year period and restoration treatments consisted on reforested plots with an induced lupine shrub cover (*Lupinus elegans*), plots with pine bark mulch and 12 plots as control.

After three years, we found significant effects of *L. elegans* on *A. religiosa* survival and growth, whereas the pine species were not affected in survival by the interaction with lupine canopies.

We found significant differences in survival between *P. pseudostrobus* plants of seven and seventeen months old (43 vs. 82% respectively). No significant effects of pine bark mulch on tree survival or growth were detected for any of the tree species tested. An additional finding was the occurrence of natural regeneration of *Crataegus Mexicana* and *Pinus pseudostrobus* under lupine canopies of legume plots which indicates a facilitating effect of the legume species. The positive effect

of *L. elegans* on *A. religiosa* establishment suggests that nurse-plant effects can be important also in temperate forests and a strategy for facilitating succession.

Keywords: ecological restoration, Nuevo San Juan Parangaricutiro, nurse plants, old fields

INTRODUCTION

Facilitative interactions among plants have been reviewed recently (Padilla and Pugnaire 2006, Brooker *et al.* 2008) because they have been found to be more frequent than previously thought. Although facilitation has been considered for a long time as fundamental for succession (Walker and del Moral 2003), several studies in the last two decades have shown that facilitative effects are also important in stable non-successional communities (Hunter and Aarssen 1998, Bertness and Callaway 1994, Callaway 1995, 1997, Brooker and Callaghan 1998). Facilitative effects seemingly increase in strength and frequency in stressful environments (Hunter and Aarssen 1998, Bertness and Callaway 1994, Callaway and Walker 1997, Brooker and Callaghan 1998).

Brooker *et al.* (2008) recently called for testing positive plant interactions as restoration tools for degraded areas. With the exception of arid environments, the role of positive plant interactions is almost always overlooked in restoration, this despite studies showing positive interactions under restoration conditions in high mountain environments (Walker and Powell 1999, Aerts *et al.* 2007), semiarid thornscrub (Jurado et al 2006), and Mediterranean vegetation (Gómez-Aparicio *et al.* 2004, Padilla and Pugnaire 2006). Facilitation has also been observed in severely degraded environments under restoration (Densmore 2005, Frérot *et al.* 2006, Padilla and Pugnaire 2006).

Several legume species have been used worldwide in restoration projects because of their nitrogen fixing properties. Nitrogen is a limiting nutrient in many temperate ecosystems (Vitousek *et al.* 2002) particularly in degraded sites (Bradshaw *et al.* 1982). Legumes usually have high growth rates, produce large quantities of high quality litter that improves soil conditions in severely degraded sites, and contribute to reestablish nitrogen cycling in restoration sites (Vázquez-Yanes *et al.* 1999; Mislevy *et al.* 1990; Ashton *et al.* 1997), and have been used to foster restoration of conifer forests (Barton 1993, Castro *et al.* 2002; Callaway *et al.* 1996; Gass and Barnes, 1998; Smit *et al.* 2006).

Shrub species, including legumes, can act as nurse plants by creating favorable conditions for the establishment of other plant species (soil structure improvement, increasing water infiltration, lower air and soil temperature, reducing soil erosion and increasing nutrient availability) (McAuliffe 1988, Keeley 1992, Cody 1993, Fisher and Gardner 1995; Sans *et al.* 1998; Chambers *et al.* 1999). Usually, shrubs already present on a site are used as nurse plants for restoration (Gómez-Aparicio *et al.* 2004) because establishing nurse plants can be time and resource consuming (Castro *et al.* 2002).

As part of a participative research effort, an adaptive restoration program was established with the Comunidad Indígena de Nuevo San Juan Parangaricutiro (CINSJP) in the state of Michoacán, Mexico. Since 2001, a series of restoration experiments was conducted, assessing the effect of *Lupinus elegans* Kunth on several ecosystem attributes. The CINSJP applies an internationally recognized sustainable forestry model that aims to produce timber and preserve biodiversity (Velázquez *et al.* 2003, Blanco-García y Lindig-Cisneros 2005). *Lupinus elegans* was chosen because it is an abundant leguminous shrub species in pine and pine–oak forests in the region, but also because it's nitrogen-fixing capability, fast growth rate and it's well known propagation requirements (Alvarado-Sosa *et al.* 2007, Medina-Sanchez and Lindig-Cisneros, 2005). In the study region, we have witnessed the dynamics of several stands of *Lupinus elegans.* The stands are established after years of mast seeding and when winter conditions are not too harsh (high survival occurs when temperatures do not fall below freezing). Established plants flower at their second growing season, when they reach maximum size (2-3 m high), but produce large quantities of seed only during the third growing season, when many plants die, so almost no plants survive the fourth growing season. Stand dynamics suggested us that this species could be used as a transient nurse-plant during the most vulnerable stages of tree species establishment, particularly for *Abies religiosa*.

To test this hypothesis, saplings of three native conifer species (*Pinus pseudostrobus, P. montezumae* and *Abies religiosa*) were planted in an agricultural abandoned field and *Lupinus elegans* was seeded as a cover treatment. The conifer species were selected for several reasons; *Pinus montezumae* and *P. pseudostrobus*, besides being the most common species in the region, are appreciated for timber production. *Pinus pseudostrobus* has a large growth potential and timber of good quality; it is produced massively across Mexico for the National Reforestation Program (CONAFOR 2008). Nurseries produce plants of two different ages at planting (7 and 17 months old). *Pinus mont*ezumae is the second economically most important species in the study area because it has a double purpose (good turpentine production and good timber quality). *Abies religiosa* is the fir species with the broadest distribution in Mexico, and is a late-successional shade-tolerant species (Velázquez *et al.* 2003, CAB 2003; CONAFOR 2008).

Plant cover and mulching treatments are widely used to improve physical soil conditions, by preventing soil drying and by reducing high soil temperatures (Barradas 2000). Pine bark mulch was selected because it has positive effects on pine survival under harsh conditions (Blanco-García and Lindig-Cisneros 2005) and it's a leftover product of local timber industry.

We began a restoration effort in collaboration with CINSJP to assess the effectiveness of *L. elegans* for improving soil conditions of abandoned agricultural lands, increase diversity through natural establishment of vascular plant species and to increase survival and growth of coniferous species through nurse-plant effects (the latter subject is addressed in this paper).

The main goal of the collaborative effort is to transform current reforestation strategies into restoration efforts by adding actions that will lead to economically productive, more diverse and structurally complex plant communities. This will also foster the establishment of late successional key species such as *Abies religiosa*. The proposed actions are focused on improving soil microclimatic conditions by using mulching or fast growing species (to create conditions for plant facilitation that will improve tree establishment).

We hypothesized that the effects of *Lupinus elegans* will be different for shade tolerant species (*A. religiosa*) and shade-intolerant species (*Pinus pseudostrobus* and *P. montezumae*), (Quintana-Ascencio et al 2004, Quintana-Ascencio and González-Espinosa M. 1993, Nieto de Pascual Pola et al. 2003). We expect positive effects for shade tolerant species through nurse-plant effects, and negative or neutral for shade-intolerant species. We also hypothesized that pine bark mulching will increase survival of conifer species by protecting the young trees.

METHODS

Study Area:

The study area is located in the CINSJP in the Northeast region of the State of Michoacán, Mexico; at 2750 m.a.s.l. Pine-oak-fir forests dominate the area. The forests of the community (11,694 ha) are managed for timber extraction under a claimed “sustainable” forestry practice. The study was carried out in a middle-slope hillside with previous agricultural land use that was abandoned 9 years before the restoration started. The surrounding landscape consists of a matrix of agricultural lands with small and dispersed fragments of pine-oak-fir forests. Despite being abandoned almost a decade ago, no tree species were present at the beginning of the study in the restoration site.

Restoration experiment establishment

In June 2004, three coniferous species (*Abies religiosa, Pinus montezumae* and *Pinus pseudostrobus*) were used to set-up a restoration experiment in an area of 4,000 m^2 (figure 1). Within the area, thirty-six 8 × 8 m plots were established with a distance of 4 m between adjacent plots, and were divided in two blocks for accounting for differences in slope across the hillside. Four conifer saplings of each species (*Abies religiosa, Pinus montezumae* and *Pinus pseudostrobus* of two ages; 7 and 17 months old) were planted inside each plot following a Latin Square design (for a total of sixteen plants per plot) with a 2 m distance between adjacent trees. Six plots in each block were selected at random and uniformly sown with 1,200 seeds of *L. elegans* in order to form a canopy of this species with an average of 1 plant per square meter. The area of each plot, $64 m^2$, is close to the average size of natural *L. elegans* clumps within the study area (Personal observation). Lupine seeds were scarified with concentrated sulphuric sulfuric acid for 30 min. prior to sowing (Medina-Sánchez and Lindig-Cisneros 2005). Six additional randomly-selected plots per block were used to test the effect of pine bark as mulching. Mulching was applied by adding a 4-cm deep layer covering 1 m^2 around each of the tree plants in the plot. The remaining 6 plots per block were kept as a control.

Conifer survival and growth were evaluated every two months during three years and four months (from June 2004 until October 2007). Also, percent cover of *L. elegans* was estimated in the quadrants during 2006 and 2007. Soil temperature was recorded hourly with data logger (Hobo® H01-001-01 Onset Computer Corporation, USA) placed in the buried 4 cm below the soil surface in one randomly selected plot of each treatment. Relative growth rates (RGR) were calculated using the following formula: RGR = (log ($h2$)-log (h_1))/ (t_2-t_1), where h_2 is the height at the end of the experiment, h_1 were initial heights and t was the time interval in years.

A follow-up experiment was set up in June 2006 to determine the effect of fully developed lupine patches on *Abies religiosa* survival and growth because most of the mortality for this species occurred during the first year of the main experiment, when lupine cover was not fully developed. Greenhouse propagated *A. religiosa* plants were replanted to replace missing individuals. The origin and age of plants used in the follow up experiment was the same of the plants used in the main experiment. In total, 105 trees were replanted: 36 in mulched plots, 31 in lupine plots and 38 in control plots).

Statistical analysis

Final conifer survival (June 2007) was analyzed using GLM for Poisson distributed data with five possible states for each plot; 0, 1, 2, 3 and 4 live plants (Dobson 2002); and two explanatory variables were explored: treatment, block and their interaction. Conifer survival through the length of the experiment was analyzed with the Kaplan-Meier estimator of survival, a function that describes the distribution of the times of survival, to further explore how the time of survival differed between legume, pine-bark mulching and control plots. Relative growth rates and height gained during the experiment were analyzed with ANOVA using the mean value for each plot as the response variable. When only one tree survived the datum was used for the analysis; when no trees survived RGR and height were considered to be zero. All analyses were carried out using S-Plus 2000 (Mathsoft 2000).

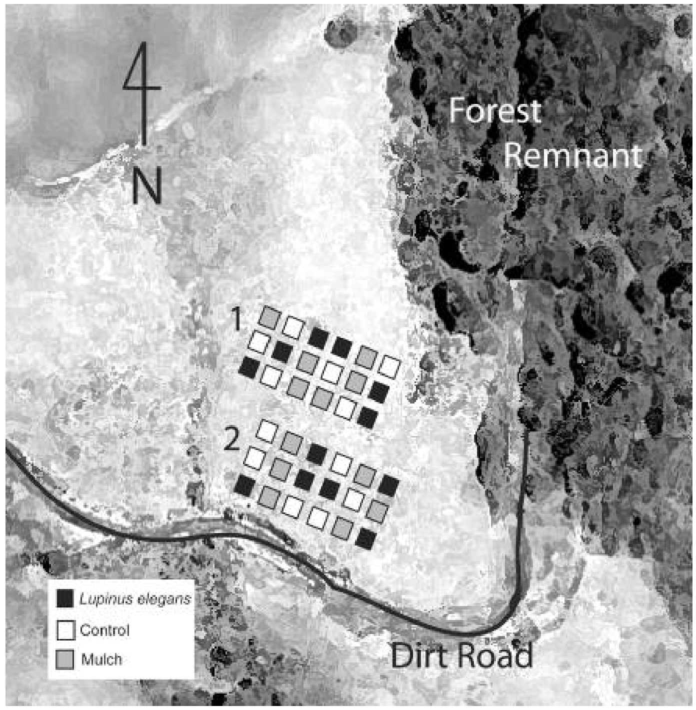

Figure 1. Experimental design of restoration treatments (blocks 1 and 2)

RESULTS

Lupinus elegans canopy development in seeded plots

Lupine canopies developed rapidly. After two growing seasons (in June 2006) *L. elegans* canopies averaged 2.2 m in height and covered from 60 to 95% of each plot. One year later, by June 2007, because of natural die-off, they covered from 40 to 65% of each plot. The site showed enormous recruitment of *L. elegans* during the rainy season of 2006 as seed production of the lupines peaked, but below-freezing temperatures caused seedling mortality to be almost of 100% by December 2006.

As *L. elegans* canopies developed they shaded the soil and noticeably reduced maximum soil temperatures (Figure 2b) when compared with control plots (Figure 2a), especially during the warmest months (April, May and June).

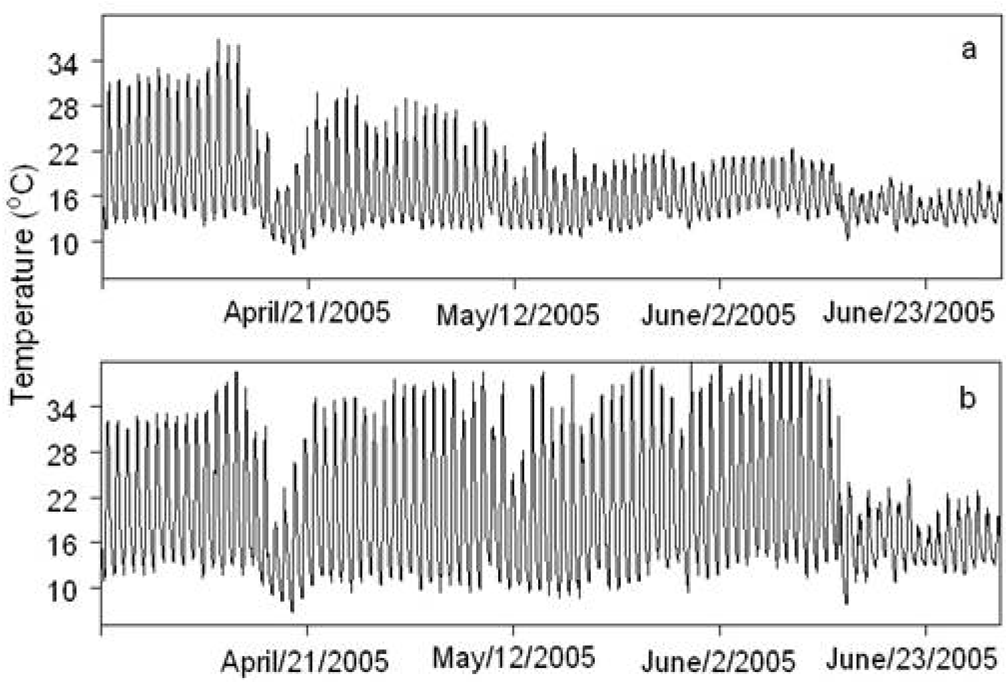

Figure 2. Air temperature during the warmest months of 2006 (April, May and June), a) *L. elegans* plots and b) control plots.

Conifer survival

Overall survival of conifer plants in the restoration experiment was 54%, and significant differences between species and treatments were found.

Lupine canopies affected differentially the survival of conifer species along the three years of this study. *Abies religiosa* plants were positively affected by the presence of *L. elegans* canopies. After three years, the treelets in legume plots survived twice as much than plants in control and mulch plots (Table 1). This trend was confirmed by GLM analysis, founding a significant effect of *L. elegans* plots on the number of *A. religiosa* plants alive per plot (Table 2). According to the Kaplan-Meier estimator, no differences among treatments were found in shape of the survival curves of *A. religiosa* during 2004-2007 (Table 3), it might be because of similar mortality during the first year of the study (which coincides with the first growing season when *L. elegans* plants reached their full size).

Pinus pseudostrobus plants showed significant differences in survival between ages at planting. Seedlings of seven months showed lower survival than seventeen months old seedlings (43% and 82%, respectively; Figure 3, Table 1). Survival of plants of both ages was higher under lupine canopies (Table 1), but the difference was not statistically significant, as shown by analysis of deviance (Table 2). A significant block effect for *P. pseudostrobus* 7 months old at planting was detected (29 vs. 57% of survival for block 1 and 2, respectively). The Kaplan-Meier estimator showed significant differences among the survival times of *P. pseudostrobus* 7 months old under different treatments (Table 3, figure 4).

Pinus montezumae showed similar survival under the treatments and blocks (Table 1).

Table 1. Percent of survival of conifer species per treatment after three years (June 2007)

species/ treatment	*A. religiosa*	*P. montezumae*	*P. pseudostrobus* 7 months old	*P. pseudostrobus* 17 months old
Mulch	8	71	33	81
L. elegans	29	75	52	92
Control	15	71	44	73
Average survival	17	73	42	83

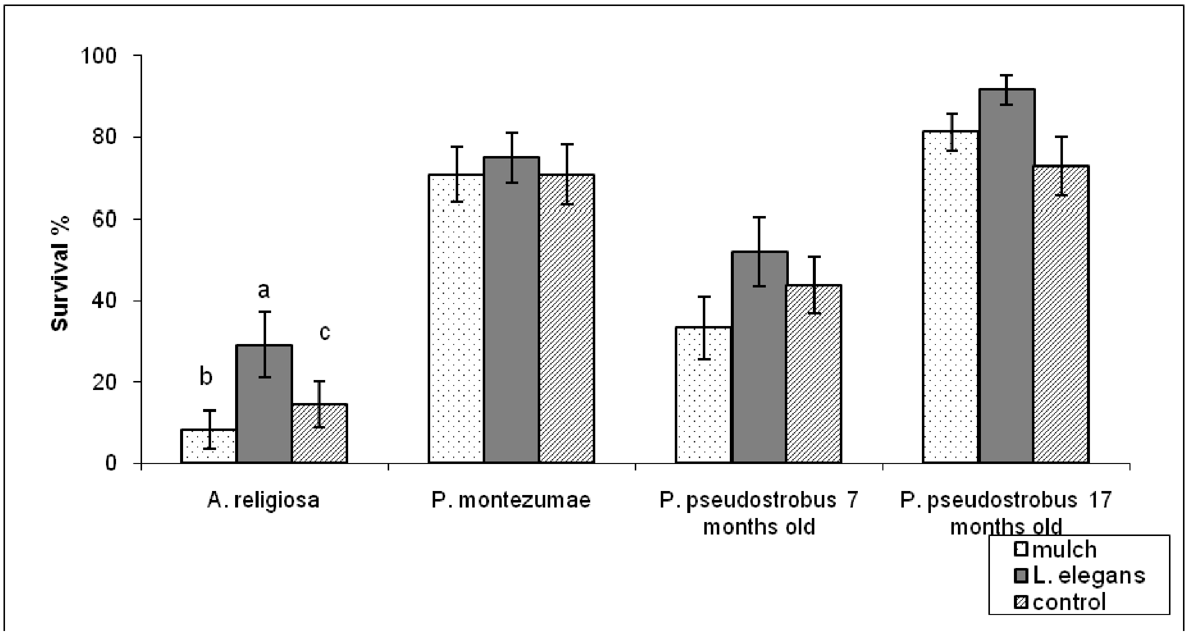

Figure 3. Survival and s.e. of conifer species per treatment after three years (June 2007)

The pine bark mulch did not have effects on survival curves or final survival per plot of conifer plants (Tables 1, 2 and 3).

Table 2. Probability values of the analysis of deviance applied to the number of survivors of conifers per plot.

A. religiosa	Df	Deviance residual	Df	Resid. Dev	P
NULL			35	45.2	
Treatment	2	6.2	33	39	0.04
Block	1	3.3	32	35.7	0.06
Treatment block	2	0.2	30	35.5	0.89

P. montezumae	Df	Deviance residual	Df	Resid. Dev	P
NULL			35	10.9	
Treatment	2	0.07	33	10.8	0.96
Block	1	0.96	32	9.9	0.32
Treatment block	2	1.41	30	8.4	0.49

P. pseudostrobus 7 months	Df	Deviance residual	Df	Resid. Dev	P
NULL			35	30.7	
Treatment	2	1.99	33	28.7	0.36
Block	1	6.56	32	22.2	0.01
Treatment block	2	0.25	30	21.2	0.88

P. pseudostrobus 17 months	Df	Deviance residual	Df	Resid. Dev	P
NULL			35	7.24	
Treatment	2	1.03	33	6.2	0.59
Block	1	0.13	32	6	0.71
Treatment block	2	0.23	30	5.8	0.88

Table 3. Results of the Kaplan-Meier estimator of survival time of each conifer species between the treatments during the three years of the study.

Species	χ^2	P
Pinus pseudostrobus 7 months	7.1	0.028 *
P. pseudostrobus 17 months	3.3	0.195
P. montezumae	0	0.997
Abies religiosa	3.3	0.196
Replanted *A. religiosa*	8.2	0.016*

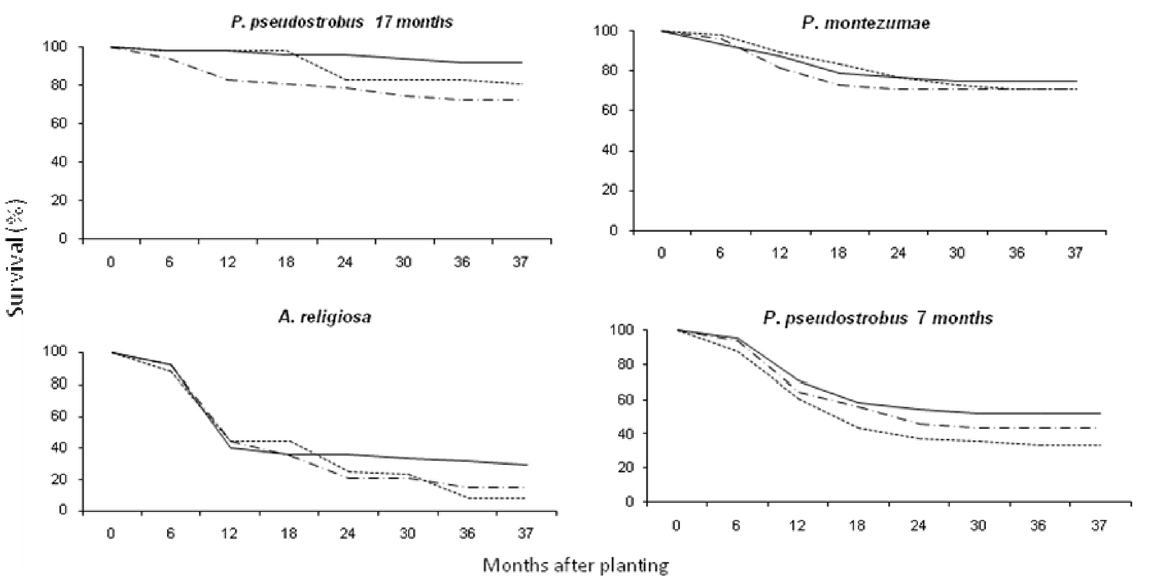

Figure 4. Survival curves of conifer species among treatments during the three years of the study: dotted line, mulching; solid line, L. elegans; dash and dot, control.

Abies religiosa plants of the follow-up experiment showed a significant treatment effect on survival times according to the Kaplan-Meier estimator (Table 3, figure 5). Lupines had a positive effect on final survival of *A. religiosa* plants when compared with mulch and control treatments (32, 8 and 10%, respectively).

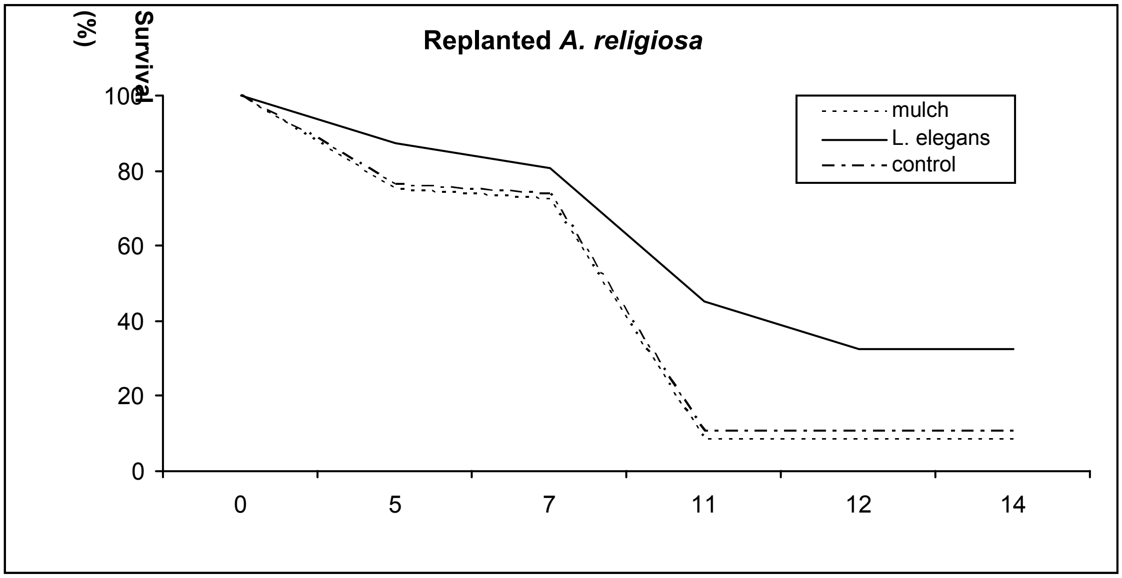

Figure 5. Survival curves of replanted *A. religiosa* plants (form July 2006 to August 2007).

Conifer growth

A significant effect of block and species was detected on relative growth rates (RGR) but no effect of treatments or any interactions was detected (Table 4). The interaction species: treatment was marginally significant because *A. religiosa* growth was higher in legume plots. In figure 6 the block effect for RGR can be appreciated for some species; being significant for *Pinus montezumae* and *P. pseudostrobus* of 7 months at planting.

Only *A. religiosa* plants showed significant differences in RGR and final height among treatments (Table 5 and 6). Tukey multiple comparisons indicate that significant differences exist between mulched and *L. elegans* plots.

Pinus montezumae and plants of both ages of *P. pseudostrobus* showed no significant differences in growth among treatments. However, it can be noticed that

P. montezumae grew less in *L. elegans* plots and *P. pseudostrobus* of 17 months grew taller on the same treatment (Table 6).

For *A. religiosa*, *L. elegans* plots had a positive effect on final heights of this shade tolerant species (Table 5 and 6), while initial heights (in 2004) were similar for all individuals among treatments, the final height (in 2007) of *A. religiosa* plants was higher in *L. elegans* plots. The differences were significant despite that only few individuals survived after three years; 3 in mulched plots, 12 under *L. elegans* canopies and 4 in control plots.

Table 4. Summary of analysis of variance with accumulated relative growth rates during 2004-2007.

	Df	F	*P*
Block	1	14.52	0.000
Species	3	90.81	0.000
Treatment	2	1.41	0.233
Block × species	3	3.53	0.017
Block × Treatment	2	0.73	0.482
Species × Treatment	6	2.13	0.057
Block × species × Treatment	6	0.45	0.843
Residuals	120		

Table 5. Summary of one way ANOVA applied to conifers relative growth rates among treatments.

	F	*p*
A. religiosa	4.31	0.02 **
P. montezumae	0.52	0.59
P. pseudostrobus 7 months	2.01	0.15
P. pseudostrobus 17 months	0.01	0.98

Table 6. Average height ± se (cm) of conifer species in 2004 and 2007

July 2004	*A. religiosa*	*P. montezumae*	*P. pseudostrobus* 7 months old	*P. pseudostrobus* 17 months old
Mulch	27 ± 1.4	9 ± 1.0	6 ± 0.4	29 ± 0.9
L. elegans	29 ± 1.6	10 ± 1.4	6 ± 0.4	29 ± 1.1
Control	27 ± 1.3	9 ± 1.3	7 ± 0.4	27 ± 1.2
July 2007				
Mulch	22 ± 3.7	99 ± 11.4	58 ± 16.0	202 ± 16.3
L. elegans	38 ± 7.7	78 ± 8.4	79 ± 15.2	216 ± 16.8
Control	22 ± 5.9	91 ± 13.3	89 ± 14.0	169 ± 21.6

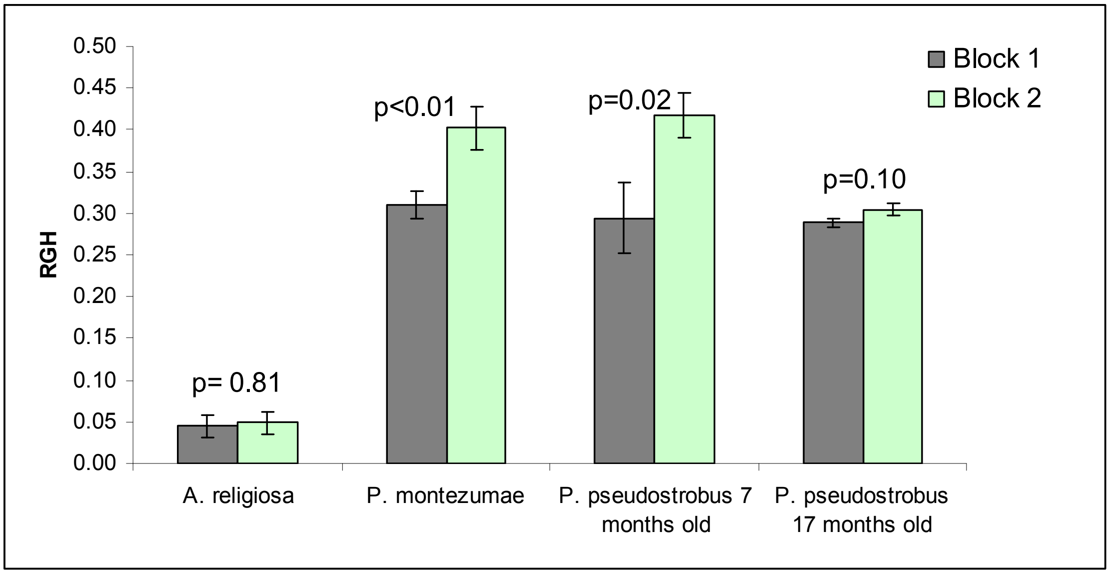

Figure 6. Conifer accumulated relative growth rates among blocks during 2004-2007 (*p* values according to a one way ANOVA)

Two tree species recruited naturally in the restoration experiment only under the canopy of lupines plots; *Crataegus* mexicana, a bird dispersed tree, and *Pinus pseudostrobus*, the most abundant conifer species in the CINSJP forests.

DISCUSSION AND CONCLUSIONS

Pinus pseudostrobus is the most abundant and economically important pine species in the CINSJP forests. Older plants of this species had better survival and growth when compared with younger plants, because a taller seedling can perform better in stressful field conditions.

Our results indicate a positive nurse plant effect for the interaction between *Abies religiosa* and *Lupinus elegans*. Current survival rates of *A. religiosa* reforestations range from 10 to 35% in the nearby Monarch Butterfly Biosphere Reserve where this conifer species is dominant in forest physiognomy (RBMM, Reserva de la Biósfera Mariposa Monarca, forest office personal communication). Our results indicate that through the nurse-plant interaction between *A. religiosa* and *L. elegans* the success of reforestations with this tree species can be improved, especially on degraded sites where the lack of tree canopies limit its survival and also for sites far from forest edges.

Since our study site is located at 2,750 m.a.s.l., in the lower limit of the natural distribution of *A. religiosa,* our restoration experiment tested the nurse effect under the most stressful conditions for this species. It might be expected that at higher altitudes survival rates and growth may be higher. This may be the case of the wintering habitat of the Monarch Butterfly (*Dannaus plexipus* Linnaeus) where *A. religiosa* forests are present at its optimal altitudinal range; between 2,900 to 3,500 m.a.s.l. (CONAFOR 2007).

This experiment did not showed negative effects of using lupines on pine species, even under the thickest *L. elegans* canopies (60 to 90% of lupine cover for each plot). *Pinus montezumae* had no response to the presence of the Lupinus plants despite being a shade-intolerant species (Gómez-Aparicio *et al.* 2004). Furthermore, reforestation practice in Mediterranean areas consider shrubs growing close to newly planted trees as heavy competitors, and consequently they are removed before planting (Gómez-Aparicio *et al.* 2004). The same practice is followed in our study site. The fact that pine species survived better and showed no disadvantages in growth under lupine plots rejects the idea that pine species cannot perform in a satisfactory way under shade conditions.

Pinus montezumae shows a grass stage during its early years of growth and it has been argued that grass stage is an adaptation to both the occurrence of frosts and wild fires (Perry, 1991).The observed higher growth of *P. montezumae* on mulched plots (although no significant) may be related to the inhibition of grasses and forbs that may otherwise compete with the trees during this initial "grass stage" characteristic of *P. montezumae*.

Our results suggest that nurse-plant effects in degraded sites are more important for late-successional shade-tolerant species than for early successional species. Temperate coniferous forests are known for following classic succession dynamics and this has been the basis for most forestry practice and restoration (Harris 1984). Coniferous forests in western Mexico can take as long as 300 years to reach late successional stages where *A. religiosa* is dominant (Yeaton et al. 1987); our results indicate that nurse plant effects can facilitate establishment of this species early in the successional process under restoration conditions.

Along the three years of the study, we documented an important increase in nitrogen content at the restoration site, including in control plots. However, lupine plots had significantly higher available nitrogen concentration than control plots, and during 2007, it began to diminish following lupine die-back. Nitrogen is a limiting nutrient in many temperate ecosystems (Vitousek *et al.* 2002) particularly in degraded sites (Bradshaw *et al.* 1982) and it may have contributed to increase the conifer survival and height at the site.

Consequences for management

Although people from the CINSJP recognize the benefits of *L. elegans* in agricultural traditional practices as a soil amendment, lupines and other shrub species are removed before reforestation because forestry practice is not based in traditional land management. Our study shows that lupines are beneficial for survival and growth of *Abies religiosa,* and are not harmful for pine species. Furthermore, lupines can facilitate natural establishment of mid-successional tree species such as *Crataegus mexicana* and *Pinus pseudostrobus.dostrobus.*

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CAPITULO 5. RESTORATION OF VASCULAR PLANT RICHNESS IN AN AGRICULTURAL ABANDONED FIELD THROUGH A NURSE-PLANT INTERACTION.

ABSTRACT

Plant richness was explored in an ecological restoration trial during three consecutive years to assess the effect of the legume *Lupinus elegans* Kunth as nurse-plant in an agricultural abandoned field. The treatments consisted in 12 plots with an induced lupine shrub cover (*Lupinus elegans*) and 12 plots as control. Plant richness was explored at plot level and also at site level by comparing the restored field and three adjacent sites: two old fields and a forest remnant. Phosphorus, nitrogen and pH were also determined and correlated to plant richness.

Plant richness increased from 18 species in 2004 to 57 in 2006 on the restored field. Lupine plots showed a significantly higher average number of vascular plant species than control plots (P <0.001), specifically on the edges of the plots. At plot level, lupine cover had greater influence on plant richness increments than available nitrogen. Furthermore, tree species recruited only under lupine plots. The two adjacent old fields remained low in plant richness (12 and 19 species) and the forest remnant showed 36 species, sharing 28 species with the restoration experiment in 2006. In 2007 the same trends were observed except for the edge of lupine plots which did not show differences in species richness with the center of the same plots.

Our results suggest that large increase in species richness in the restored field, when compared with adjacent areas, is a consequence of spatial heterogeneity created by lupines. The occurrence of seedlings of the tree species, *Crataegus mexicana* and

Pinus pseudostrobus Lind in the restoration field indicates the potential of *L. elegans* to improve natural establishment of native tree species.

Keywords: ecological restoration, Nuevo San Juan Parangaricutiro, nurse plants, old fields, tree plantations, diversity.

INTRODUCTION

When rural communities shift land use from agriculture to sustainable forestry, reforestation is a common practice, but commonly leads to monotypic forest plantations (Newmaster *et al.* 2006). Forest plantations have increased at exponential rates worldwide but have been criticized because these monocultures are "biological deserts" and also because of the use of exotic species (Lugo 1997). Recently, the need to manage plantations to accelerate their conversion to more natural forest types, to increase biodiversity and ecosystem services, has been noted (Keenan *et al.* 1997; Newmaster *et al.* 2006). When communities shift from non-sustainable forestry or from intensive agricultural practices to sustainable management, restoration efforts require practices allowing native forests to recover (Sarr *et al.* 2004). Restoration becomes even more relevant when maintaining biodiversity is a concern, a requirement for sustainability certification.

In some cases, the aim of forest management has changed from a single objective (wood production) to multiple objectives (carbon sequestration, habitat for wildlife and native plants, and control of soil erosion). However, the contribution of reforestation to sustaining biodiversity is unclear. Studies on how to increase biodiversity in reforestations for timber production are scarce in the discipline of restoration ecology (Newmaster *et al.* 2006; Nagaike 2002), despite the fact that the recovery of species composition and structure is the most fundamental restoration goals (Cairns 2000). Furthermore, biological diversity in ecosystems, including agro-ecosystems, provides a range of biological functions, such as nutrient cycling and pest control (Altieri 1999; Marshall *et al.* 2003) that are fundamental for the long term viability of the restored site (Tillman *et al.* 1996; Naeem *et al.* 1994; Wilsey and Potvin 2000; Lindig-Cisneros and Zedler 2002; CBD 1992).

Several legume species have been used worldwide in restoration projects because of their capability for nitrogen fixation. Nitrogen is a limiting nutrient in many temperate ecosystems (Vitousek *et al.* 2002) particularly in degraded sites (Bradshaw *et al.* 1982). Legumes usually have high growth rates, produce large quantities of high quality litter that improves soil conditions in severely degraded sites, and contribute to reestablishing nitrogen cycling in restoration sites (Vázquez-Yanes *et al.* 1999; Mislevy *et al.* 1990; Ashton *et al.* 1997). Nevertheless, nitrogen saturation can lead to soil acidification, establishment of undesired weed or invasive species, and higher productivity which is coupled with lower species richness under certain conditions (Maron and Connors 1996; Maron and Jeffries, 2001; Goldberg and Miller, 1990; Prober *et al.* 2005).

Shrub species, including legumes, can act as nurse plants by creating favorable conditions for the establishment of other plant species (McAuliffe 1988, Keeley 1992, Cody 1993, Fisher and Gardner 1995; Sans *et al.* 1998; Chambers *et al.* 1999), and have been used to foster the establishment of conifer species in ecological restoration (Barton 1993, Castro *et al.* 2002; Callaway *et al.* 1996; Gass and Barnes, 1998; Smit *et al.* 2006). Usually, shrubs already present on site are used as nurse plants for restorations (Gómez-Aparicio *et al.* 2004) because establishing nurse plants can be time and resource consuming (Castro *et al.* 2002).

As part of a participative effort an adaptive restoration program was established with the Comunidad Indígena de Nuevo San Juan Parangaricutiro (CINSJP) in the state of Michoacán, Mexico. Since 2001, a series of restoration experiments was conducted assessing the effect of *Lupinus elegans* Kunth on several ecosystem attributes, including the performance of native conifer species. The CINSJP applies an internationally recognized sustainable forestry model

(Velázquez *et al.* 2003, Blanco-García y Lindig-Cisneros 2005). *Lupinus elegans* was chosen because it is an abundant leguminous shrub species in pine and pine-oak forest in the region, but also because it's nitrogen-fixing capability, fast growth rate and its well known propagation requirements (Alvarado-Sosa *et al.* 2007, Medina-Sanchez and Lindig-Cisneros, 2005). In the study region, we have witnessed the dynamics of several populations of this species because restoration progress is monitored every two months at different experimental sites

We began a restoration experiment in collaboration with CINSJP to assess the effectiveness of *L. elegans* for improving soil conditions of abandoned agricultural lands, increase survival and growth of coniferous species through nurse-plant effects, and increase diversity through the establishment of vascular plant species. Effects on vascular-plant species richness at different scales are reported here. We evaluated the effects on vascular-plant species richness at the scale of the nurse plant, as have been done on most studies on nurse-plant effects (Carrillo-García *et al.* 1999), but we also evaluated effects at the site scale created by the heterogeneity of lupine canopies.

We hypothesized *Lupinus elegans* will increase species richness by improving site conditions (creation of microsites for germination) through nurse-plant effects, and nitrogen enrichment will not affect negatively species richness, given the very low concentrations characteristic of degraded agricultural lands in the region (between 95 and 113 kg/ha of total nitrogen. Also, because of the considerable height that this species can reach after the second growing season, we hypothesized bird-dispersed species will be more abundant in restored areas with *L. elegans* than in areas without them.

METHODS

Study Area:

The study area is located in the CINSJP in the Northeast region of the State of Michoacán, Mexico at 2750 m above sea level. Pine and oak–pine forests dominate the area. The forests of the community (11,694 ha) are managed for timber extraction under sustainable forestry practice (Velazquez *et al*. 2003). The study was carried out in a middle-slope hillside with previous agricultural land use and was abandoned at least 9 years ago. The surrounding landscape consists of a matrix of agricultural lands with small and dispersed fragments of pine-oak-fir forest. Despite being abandoned almost a decade ago, our study site showed a low number of plant species, both forbs and weeds, and no tree species were present at the beginning of the study.

Restoration experiment establishment

In June 2004, three coniferous species were used to reforest a 1.15 ha hillside (Fig 1). A restoration experiment was set-up in 0.45 ha of the hillside and three coniferous species were planted: *Pinus pseudostrobus* Lind, *Pinus montezumae* A. B. Lambert and *Abies religiosa* (Kunth) Schltdl. & Cham (Area A in figure 1), in a square matrix with 2 m between neighboring trees. In the rest of the site only the pine species were planted. Within area A, thirty-six 8 × 8 m plots were established with a distance of 4 m between adjacent plots, and divided in two blocks. Twelve plots, selected at random, were uniformly sown with 1,200 seeds of *L. elegans* in order to form a canopy of this species with an average of 1 plant per square meter. The area of each plot, 64m^2, is close to the average size of natural *L. elegans* clumps in the study area (Personal observation). Twelve additional randomly-selected plots were used to test the effect of mulching (these plots are not considered in this study) and the remaining 12 plots were maintained as a control. After one growing season (in June 2005), *L. elegans* canopies averaged 2.2 m in height and covered from 60 to 95 % of each plot.

From June to October 2004, a detailed survey of vascular plant richness was carried out on the hillside before reforestation and the same survey was made in 2006 and 2007 (the survey consisted in registering the whole plant richness). Besides the restoration experiment (the experimental plots, A in figure 1), we surveyed three additional areas of the hillside: 1) a 3,500-m^2 area on the top of the hillside, B in figure 1; 2) a 3,500 m^2 area adjacent to the experimental plots, C in figure 1; and 3) a pine-oak forest remnant adjacent to our restoration site, all areas sampled with the same criteria (registering all vascular plant species in each area). The maximum distance between the four sites is less than 150 m.

Besides the survey at site level, we explored vascular plant richness at two additional scales: 1) lupine and control plots; and 2) inside lupine plots considering both the central area of 36 m^2 and the edge of each plot of 64m^2.

Specimens of all the plants species found were collected and identified. Soil samples were collected in 2004, 2006 and 2007 (one sample per plot) and were analyzed by a specialized laboratory (CBTA Soil Laboratory, Morelia, Michoacán) .

Analysis of variance was used to compare soil acidity (pH values), phosphorus and nitrogen between lupine and control plots, and analysis of covariance to test the relationship among plant richness and several other variables. At the site level, we explored Beta diversity with presence/absence data applying Jaccard's similarity index. Analyses were carried out with S-Plus (Mathsoft, 2000).

RESULTS

A total of 66 and 78 vascular-plant species were found in our study site in 2006 and 2007 respectively. Plants from the Asteraceae and Poaceae families were the most abundant (see Appendix). Most species found were native, including some endemics; few exotic species (naturalized according to some authors, CONABIO 2008) like *Taraxacum offcinale* L. and *Raphanus raphanistrum* L. *were* recorded.

At site level, vascular-plant richness showed a significant increase (39 new species) in our restoration experiment (A in figure 1) after two years. The other two sites (B and C) maintained low species richness in 2006 (12 and 19 species, respectively), similar to the hillside in 2004 (18 species), whereas in the forest edge 36 species were registered (Table I), and very similar patterns were recorded en 2007 for species richness on the four sites.

The Jaccard's Similarity Index calculated with presence-absence data shows that areas B and C were the most similar (0.66), followed by the forest edge and our restoration experiment (A) in 2006 (0.59) which shared 28 species. Area B in 2006 was dissimilar from 2004 (0.32) as well as from the restoration site (area A) in 2006 (0.32), (Table I).

In 2006, thirty species (45%) were exclusive to one site (21 species on the restoration experiment and 9 species on forest edge), while areas B and C showed no exclusive species. Six species (*Commelina tuberosa* L., *Dalea tohuinii* Schrank, *Hypericum philonotis* Schltdl. & Cham., *Trifolium mexicanum* Hemsl., *Bidens aurea* (Aiton) Sherff and *Bidens serrulata* Poir. Desf.) were recorded in all sites including the restoration site before and after seeding *L. elegans*. On the other hand, species like *L. elegans* Kunth*, Salvia mexicana* L.*, Tagetes micrantha* Cav.*, Jaegeria hirta* (Lag.) Less. and *Crusea longiflora* Wild. ex Roem. & Schult. were only recorded in two sites: the restoration experiment and the forest edge.

The restoration experiment showed enormous recruitment of *L. elegans* during the rainy season of 2006 as seed production of the mature lupines peaked, but below-freezing temperatures caused seedling mortality to be almost of 100 % by December 2006. Moreover, two tree species (*Crataegus mexicana* Moc. & Sessé ex DC., a bird dispersed tree, and *Pinus pseudostrobus*, the most abundant conifer in the CINSJP forests) recruited only in the restoration experiment, only under the canopy of lupines (*C. mexicana* occurred en six legume plots and *P. pseudostrobus* in two legume plots)

At the plot level, lupine plots showed a significantly higher average number of vascular plant species than control plots in 2006 (P <0.001, Figure 2) but no differences were observed in 2007.

The analysis of covariance with 2006 data showed that lupine canopy cover influenced plant richness within plots (P=0.0001), whereas available N had no significant effect (Table II).

Some plant species showed preference for one treatment; *Jaegeria hirta* and *Plantago australis* Lam. appeared less frequently in control plots; whereas *Trifolium amabile* Kunth.*, Cyperus manimae* Kunth and *Senecio salignus* DC. were recorded more frequently in control plots. Some species were recorded in almost all plots, showing no relationship to any treatment, in particular; *Bidens serrulata, Conyza schiedeana* (Less.) Cronquist*, Dalea tohuinii, Commelina tuberosa* and *Festuca amplissima* Rupr.

Within the plot level, in 2006 the edge of lupine plots showed higher average number of vascular plant species than the center of these plots, being the difference marginally significant (P =0.07, Figure 3), but no differences were found in 2007. However, *Oenothera pubescens* Willd. ex Spreng. was more frequently recorded on the edges of lupines plots, whereas *Plantago australis* was observed only in center of lupine plots. Both *Trifolium* species were scarcely registered on the center of plots while they were abundant on the edges.

Available N increased in the whole restoration experiment (area A) from 2004 to 2006, including control plots. However, lupine plots had significantly higher available nitrogen concentration than control plots, and during 2007, it began to diminish according to lupine die-back (P =0.004, Figure 4). Other elements, including available P, decreased in the restoration experiment from 2004 to 2006 and the average concentration was significantly lower in control plots than lupine plots ($P<$ 0.01), (Figure 5), and P contents increased also during 2007. Despite these changes, the analysis of covariance (table 4) showed that N does not influence species richness and only lupine canopy significantly explains species richness in our site.

The pH values show that both leguminous and control plots have acidified since 2004 ($P<$ 0.01), but during the last year control plots maintained the same value to 2006 while lupine plots have a lower pH (Figure 6).

DISCUSSION AND CONCLUSIONS

Our results suggest that the large increase in species richness in the restoration site when compared with adjacent reforested areas was a consequence of spatial heterogeneity created by lupines canopies in a matrix of high solar radiation (creation of zones with different levels of sunlight exposure, pH, N and P). Microsites for germination around lupine plots and the different contents of nitrogen and phosphorus among treatments may have allowed species with different life histories and physiological requirements to colonize the site.

Some studies have reported a decrease in species richness after nitrogen enrichment; Maron and Connors (1996) reported a reduction in average species richness of 47% in plots where *Lupinus arboreus* had recently died (with former increasing of nitrogen content) with respect to the control plots; Carson and Barret (1988) found an increase in species richness in the first year after nutrient enrichment plots but it was lower thereafter; Eschen *et al.* (2007) registered a reduction of NO_3^- in abandoned fields through carbon addition, which also reduced above ground biomass, leading to the partial replacement of grass species by forbs and an increase in species richness. What remains clear is that the sites above mentioned experienced severe nitrate enrichment, and despite nitrogen increased from 2004 to 2006 at our restoration experiment, its content kept on intermediate levels when compared with sites with high nitrogen concentrations, and showed an opposite trend in regards to species richness. It is important to mention that we do not expected N saturation at our site for many reasons; first, *L. elegans* is a perennial short-lived species that dies after its third year of age, consequently N fixation will not continue; second, even though *L. elegans* is abundant in the seed bank, its germination and recruitment are limited by abiotic factors (temperatures of -8 ºC dropped survival of seedlings to less than 1% in 2006-2007 winter at the restoration site); and third, each control and lupine plot holds a mix of 16 conifers that are currently between 2 and 3-m tall. Since overstory biomass (a factor absent on studies previously mentioned) has been shown to uptake 10-15 times more N than understory biomass (Corbin *et al.* 2003), we expect these conifers to assimilate most of the available N and reduce the likelihood of N saturation at this site.

The acidification process observed from 2004 to 2007 is commonly associated with the use of legumes (Eschen *et al.* 2007; Corbin *et al.* 2003) but it is surprising that control plots acidified more than lupine plots in 2006, although this trend was reverted in 2007 because control plots maintained the same pH values, and lupine plots became more acid.

With respect to the risk of legumes promoting weed invasion, as they did in coastal sage scrub in California (Maron and Connors 1996; Maron and Jeffries, 2001), we did not register as many introduced species as they did (50% versus less than 3% in our restoration site), and most of the species we found are native agricultural weeds already present at the site at the beginning of the study. The low number of introduced species at our site is surprising given the surrounding landscape (mostly abandoned fields), although the site is fairly remote, and distant from highways that are traditional pathways for the spread of exotics.

The occurrence of species like *Salvia mexicana, Solanum lanceolatum* Cav., *Tagetes micrantha* Cav.*, Jaegeria hirta* and *Crusea longiflora,* only in our restoration experiment and in the forest edge, suggests that lupine canopy facilitates the establishment of plants characteristic of forest borders and the forest interior. Moreover, the occurrence of *Crataegus mexicana* and *Pinus pseudostrobus* Lindl. in the restoration experiment shows the potential of *L. elegans* to improve natural establishment of native tree species.

A factor that might have influenced the lower species richness under the lupine closed canopies in 2006 is reduced light penetration and higher litter accumulation than in border or control plots. Even though the center of lupine plots had higher amounts of total N, light interception could limit species richness through the inhibition of shade intolerant plants (Goldbeg and Miller 1990).

The higher species richness recorded on lupine plot borders could be explained by intermediate levels of sunlight, with control and lupine plots as the extreme points of the gradient (full exposure and severe shadow respectively). During 2007 *L. elegans* shrubs start to die and opened canopies might explain the similar richness values recorded between the treatments.

As a consequence of high concentrations of available nitrogen in the hillside, plants growing under lupine canopy were bigger than plants of the same species in control plots. The decrease in phosphorus concentrations after two years might be explained by the wider spectrum of species that colonized the hillside, including sowed lupines which used the nutrient during the N biological fixation (Corbin *et al.* 2003).

Consequences for management

Although people from the CINSJP recognize the use of *L. elegans* in agricultural traditional practices, and the possible benefits of the use of legumes as a soil amendment, they mow them before reforestation is implemented because forestry practice is not based in traditional land management but in technical principles. Our study shows that the use of lupines can aid in the establishment of mid-succesional species such as *Crataegus mexicana* and *Pinus pseudostrobus* and increase biodiversity, an important asset for sustainable certification and for maintaining ecosystem services. Underway monitoring has shown that lupines also are beneficial for survival and growth of planted trees in reforestation experiments.

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SPECIES	FAMILIA	RE O4	RE 06	RE 07	FE 06	FE 07	T 06	T 07	AH 06	AH 07
Cirsium ehrenbergi Sch. Bip.	Asteraceae					X				
Fuchsia microphylla Kunth	Onagraceae					X				
Acaena elongate L.	Rosaceae			X		X				
Adiantum andicola Liebm.	Pteridaceae				X	X				
Aegopogon cenchroides Humb. & Bonpl. ex	Poaceae			X	X					
Alchemilla procumbens Rose	Rosaceae		X	X						
Alchemilla sibbaldieafolia Kunth	Rosaceae	X	X	X	X					
Alchemilla velutina s. watson	Rosaceae		X	X		X		X		X
Aster subulatus michx.	Asteraceae		X	X	X	X		X		X
Bacharis heterophylla Kunth.	Asteraceae		X	X		X				
Berberis moranensis chult.& chult. F.	Berberidaceae		X	X						
*Bidens aurea(*aiton)sherff	Asteraceae	X	X	X	X	X	X	X	X	X
Bidens bigelovi A. Gray	Asteraceae	X	X	X	X	X		X		X
*Bidens serrulata(*poir.) desf.	Asteraceae	X	X	X	X	X	X	X	X	X
*Brassica nigra(*L.)W.D.G.koch	Brassicaceae		X	X	X					
Chloris sp sw.	Poaceae		X	X						
*Cologania broussonetti (*balb.) DC	Fabaceae				X					
Commelina tuberosa L.	Commelinacea	X	X	X	X	X	X	X	X	X
Conyza coronopifolia kunth	Asteraceae	X	X	X		X				X
Conyza schiedeana (less.) cronquist	Asteraceae	X	X	X	X			X		
*Crataegus mexicana l*oudun	Rosaceae		X	X						
Crusea longiflora (willd.Ex roem.& chult)	Rubiaceae		X	X	X	X		X		X
Cuphea sp. P.browne	Lythraceae		X	X	X	X	X		X	
Cyperus manimaekunth	Cyperaceae		X	X		X	X	X	X	X
Dalea thouinii Schrank	Fabaceae	X	X	X	X	X	X	X	X	X
Drymaria malachioides briq*.*	Caryophyllacea		X	X						
Eupatorium pazcuarense kunth	Asteraceae		X	X	X	X	X		X	
Festuca amplissima rupr.	Poaceae	X	X	X	X	X		X	X	X
Gallium aschenbornii Neck	Rubiaceae		X	X						
Gnaphalium ttenuatum DC.	Asteraceae	X	X	X	X	X		X		X

SPECIES	FAMILIA	RE O4	RE 06	RE 07	FE 06	FE 07	T 06	T 07	AH 06	AH 07
Gnaphalium americanum Miil.	Asteraceae		X	X	X	X			X	X
Hypericum philonotis Schltdl. & Cham.	Clusiaceae	X	X	X	X	X	X		X	X
Hypoxis sp.	Hipoxaceae		X	X	X	X	X			
*Jaegeria hirta (*lag.) less.	Asteraceae		X	X	X	X				X
*Lepechinia caulescens(*ortega) epling	Labiatae		X	X						
Lopezia racemosa cav.	Onagraceae			X	X	X				
Lotus repens (G. Don) Sessé & Moc. ex	Fabaceae		X	X	X					
Lupinus elegans kunth	Fabaceae		X	X	X	X				
Muhlenbergia macroura (kunth) hitchc	Poaceae		X	X		X		X		X
Muhlenbergia minutissima(steud) swallen	Poaceae				X	X				X
Muhlenbergia pusilla steud	Poaceae		X	X					X	
*Muhlenbergia virletti*E. Foun) soderstr	Poaceae		X	X					X	
Oenothera pubescens Wild. Ex Spreng.	Onagraceae	X	X	X	X	X			X	X
Oxalis sp.	Oxalidaceae		X	X	X					
Phacelia platycarpa (Cav.) Spreng.	Hydrophyllacea	X	X	X	X	X		X	X	X
Phaseolus sp L.	Fabaceae				X	X				
Physalis volúbilis waterf	Solanaceae		X	X						
Pinus pseudostrobus Lindl	Pinaceae		X	X						
Plantago australis lam.	Plantaginaceae		X	X						
Prunella vulgaris L.	Labiatae		X	X					X	
Raphanus raphanistrum L.	Brassicaceae		X	X				X	X	X
Sabazia humilis (kunth) cass	Asteraceae	X	X	X	X	X	X			
Salvia mexicana L.	Labiatae		X	X	X	X				
Senecio mexicanus mc vaugh	Asteraceae		X	X		X				
Senecio salignus DC.	Asteraceae	X	X	X			X	X	X	
Senecio stoechodiformis DC.	Asteraceae		X	X						
Solanum lanceolatum Cav.	Solanaceae		X	X						

SPECIES	FAMILIA	RE O4	RE 06	RE 07	FE 06	FE 07	T 06	T 07	AH 06	AH 07
sp 25			X	X						
sp 26					X	X				
sp 30				X	X	X				
sp 31					X					
sp 4			X	X						
sp 7			X	X						
Tagetes micrantha *cav.*	Asteraceae		X	X	X	X				
Taraxacum officinale F.H. Wigg.	Asteraceae		X	X						
Tithonia tubiformis (Jacq.) Cass.	Asteraceae	X	X	X	X	X			X	
Trifolium amabile Kunth	Fabaceae	X	X	X						
Trifolium mexicanun hemls	Fabaceae	X	X	X	X	X	X	X	X	X
Zeugites americanus Willd.	Poaceae		X	X						
TOTAL		18	57	61	37	36	12	17	19	21

APENDIX. Species recorded on the different sites

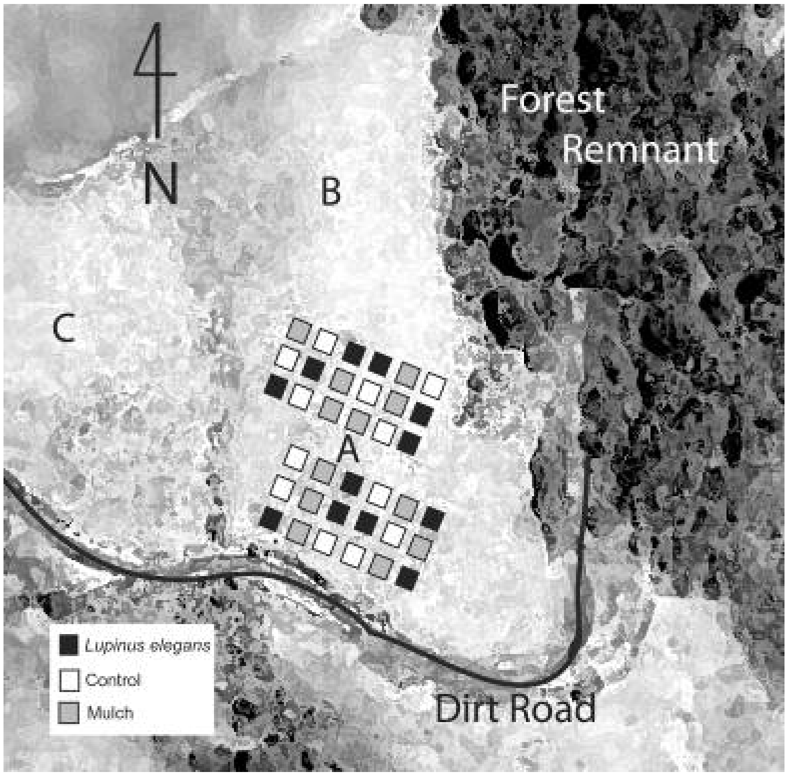
SITES: RE = restoration experiment, FE= forest edge, T= top, AH= adjacent hillside,

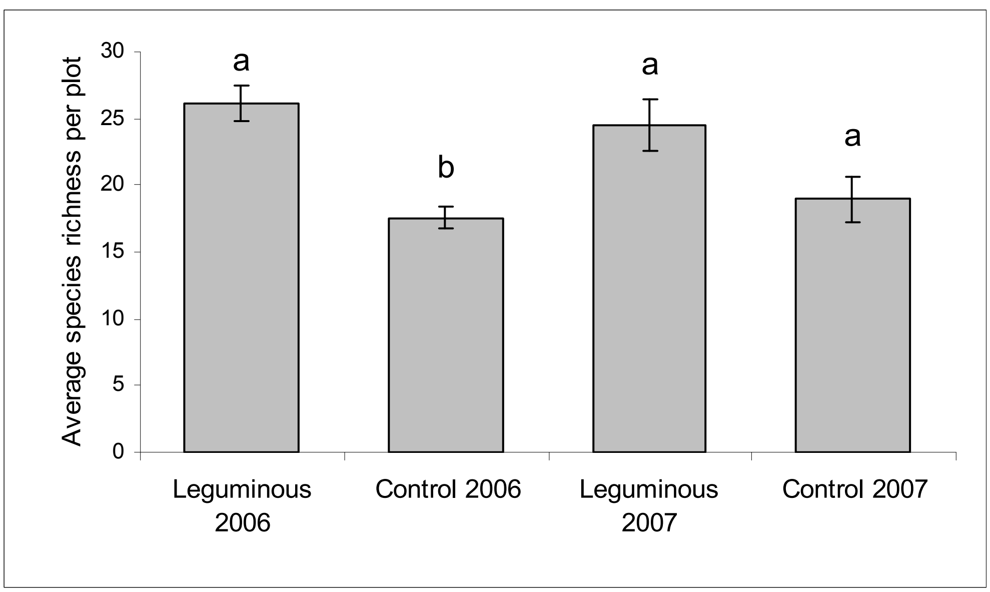
Status and habitat of the species from Medina *et al.* 2000; McVaugh, 1984; Malezas de Mexico, 2007.

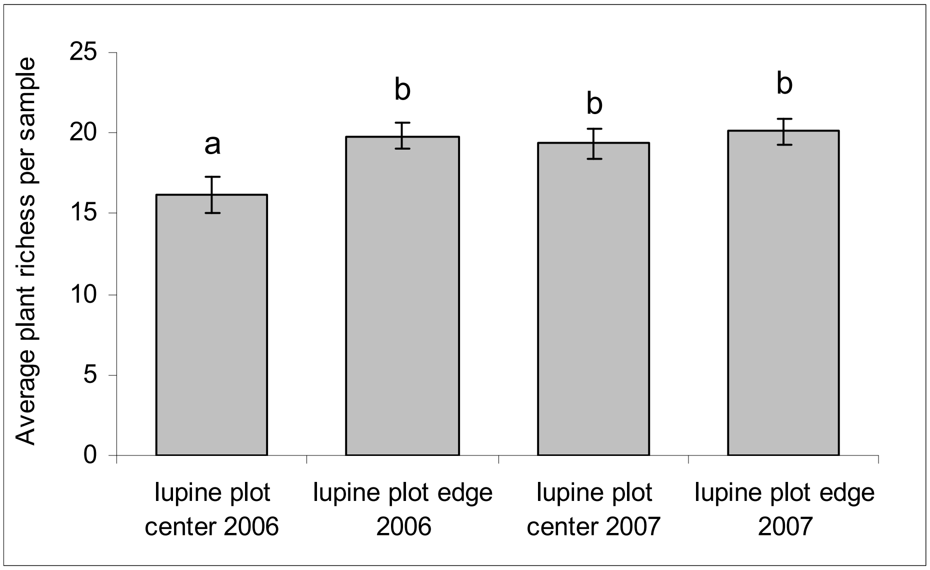
LIST OF TABLE AND FIGURES

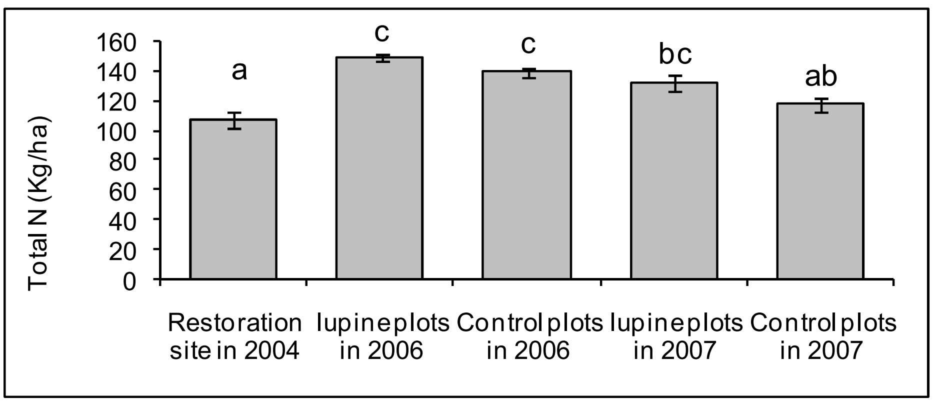
	Number of species in 2006	Area A (Restoration Experiment)	Area B	Area C	Forest remnant	Hillside in 2004	**Number of species in 2007**
Area A (Restoration Experiment)	**57**	1					**64**
Area B	**12**	0.323	1				**17**
Area C	**19**	0.5	0.666	1			**21**
Forest remnant	**37**	0.595	0.375	0.464	1		**36**
Hillside in 2004	**18**	0.394	0.32	0.545	0.431	1	

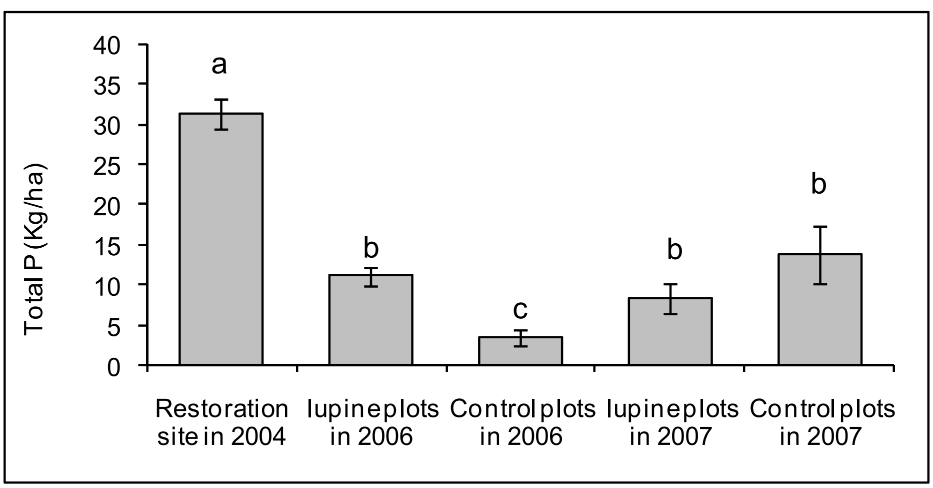
	df	Sum of Sq	Mean Sq	F	Pr (F)
Available N	1	42.79	42.79	2.31	0.14
Lupine cover	1	407.40	407.40	22.03	0.0001
Treatment	1	2.47	2.47	0.13	0.71
Available N: Lupine cover	1	18.69	18.69	1.01	0.32
Residuals	19	351.24	18.48		

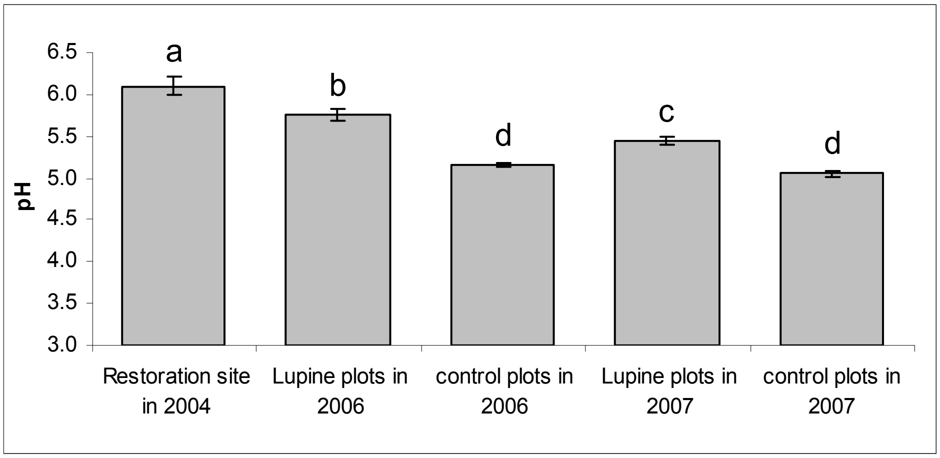

N
Forest
Remnant
B
C
A
Dirt Road
Lupinus elegans
Control
Mulch


Average species richness per plot
30
25
20
15
10
5
0
a
b
a
a
Leguminous 2006
Control 2006
Leguminous 2007
Control 2007


Average plant richness per sample
25
20
15
10
5
0
a
b
b
b
lupine plot center 2006
lupine plot edge 2006
lupine plot center 2007
lupine plot edge 2007


Total N (Kg/ha)
160
140
120
100
80
60
40
20
0
a
c
c
bc
ab
Restoration site in 2004
lupine plots in 2006
Control plots in 2006
lupine plots in 2007
Control plots in 2007


Total P (Kg/ha)
40
35
30
25
20
15
10
5
0
a
b
c
b
b
Restoration site in 2004
lupine plots in 2006
Control plots in 2006
lupine plots in 2007
Control plots in 2007



pH
6.5
6.0
5.5
5.0
4.5
4.0
3.5
3.0
a
b
d
c
d
Restoration site in 2004
Lupine plots in 2006
control plots in 2006
Lupine plots in 2007
control plots in 2007

CONCLUSIONES GENERALES

En la reciente teoría desarrollada para la Ecología de la Restauración, destaca la relación entre el nivel de perturbación y el esfuerzo de restauración propuesta por Hobbs y Norton (1996) ya que en sitios con niveles altos de perturbación se requiere de medidas aditivas tendientes a modificar las condiciones del sitio, lo que implica incrementar el esfuerzo de restauración (número y tipo de acciones tomadas). También el modelo de espectro de restauración de Zedler (1999) menciona que a mayor perturbación se requiere más de una acción sobre un solo factor limitante de la restauración. La relación entre estas dos variables (nivel de degradación y esfuerzo de restauración) no es necesariamente lineal, puede ser exponencial, haciendo difícil la proyección de resultados esperados en los proyectos de restauración. Este hecho es particularmente interesante ya que los proyectos de restauración son frecuentemente visualizados como una cuestión de manejo y aceleración del proceso de sucesión ecológica, sin embargo las condiciones presentes en sitios bajo restauración suelen ser bastante diferentes a las que tienen lugar en un sitio que se está recuperando de una perturbación natural (Hilderbrand *et a.l* 2005).

En el modelo de espectro de restauración que aquí se propone y que fue construido a partir de los resultados de Nuevo San Juan se puede observar que los tres sitios evaluados difieren notoriamente en la naturaleza e identidad de las barreras a la restauración y por consiguiente algunos presentan mayor dificultad que otros para trabajar en su recuperación

El campo agrícola abandonado (San Nicolás) presenta un grado de disturbio moderado (pérdida de cobertura forestal y la diversidad del sotobosque, así como disminución de la fertilidad del suelo) y sugiere ser un sitio que requiere un menor esfuerzo para intentar escenarios de restauración *sensu stricto;* es decir tratar de restaurar una composición y estructura de especies muy similar a la original, así como aspectos funcionales como mejorar la fertilidad del suelo. Los objetivos anteriores podrían alcanzarse de la siguiente manera;

- Posibilidad de incidir en una mayor riqueza de especies arbóreas (diversidad en reforestación) ya que las condiciones del sitio permiten el uso de más de una especie (al menos *A. religiosa, P montezumae* y *P. pseudostrobus* pueden establecerse exitosamente atendiendo algunas recomendaciones desprendidas en los capítulos revisados anteriormente.
- Posibilidad de incidir en una mayor riqueza del sotobosque mediante el sembrado de *L. elegans,* favoreciendo así la regeneración de especies arbustivas (incremento de la diversidad por nodricismo).
- Contribución al restablecimiento de ciclos biogeoquímicos mediante el sembrado de *L. elegans* (incremento de los contenidos de nitrógeno) para mejorar la productividad forestal.

El arenal que fue sujeto a extracción (Llano de Pario) muestra un grado de deterioro elevado, pero al retirar el principal agente de disturbio (una capa de arena volcánica de aproximadamente metro y medio de grosor) se incrementa notablemente la el éxito de las reforestaciones, ya que se disminuye la temperatura promedio del suelo en los meses más cálidos del año. Sin embargo las bajas temperaturas durante el invierno siguen siendo un factor ambiental crítico para la restauración de este sitio (la topografía particular del sitio recrudece el efecto de las heladas sobre las reforestaciones).

Por lo tanto para este sitio se han encontrado técnicas dirigidas a conformar una cobertura arbórea inicialmente monoespecífica de *P. montezumae* resistente a las heladas (restauración *sensu lato*) ya que actualmente, ante la baja cobertura arbórea, muy pocas especies arbóreas introducidas mediante reforestaciones podrían progresar en las condiciones climáticas del sitio.

La dinámica de las perturbaciones naturales es modificada por las perturbaciones antrópicas, creando un mosaico en los ecosistemas que comprende diversos estados, algunos de ellos estables o cuasi estables. En las tierras de Nuevo San Juan Parangaricutiro, la transición de terrenos agrícolas a bosques se da cuando se abandonan las prácticas agrícolas que mantienen el estado y es auxiliada por la reforestación, particularmente con *P. pseudostrobus*. En la transición de sitios agrícolas a bosque maduro, de acuerdo a experimentos llevados a cabo hasta la fecha, dos barreras principales limitan el establecimiento de las especies arbóreas, las heladas y la actividad de forrajeo de mamíferos pequeños (Blanco-García y Lindig-Cisneros 2005), los cuales pueden eliminar a un alto porcentaje de las plantas (hasta un 70% en el lapso de un año). Sin embargo, la regeneración natural puede generar la transición entre estos dos estados alternativos (Lindig-Cisneros *et al*. 2007).

En el caso del arenal remanente de la erupción volcánica (Cutzato, capítulo 3), la medida de restauración que ha resultado más eficiente para incrementar la supervivencia y el desempeño (evaluado como crecimiento) de *Pinus pseudostrobus* ha sido el uso de acolchados. El uso de acolchados puede llegar a duplicar la supervivencia de esta especie en años secos (Blanco-García y Lindig-Cisneros 2005). La protección contra el daño mecánico causado por la escorrentía y la mortalidad causada por la herbivoría también permiten una mayor supervivencia de esta especie y de otras como *Lupinus elegans* (Blanco-García y Lindig-Cisneros 2005). Los diversos experimentos que hemos realizado nos permiten llegar a la conclusión de que este es el estado que más barreras presenta para transitar hacia bosques. Las principales barreras que se han identificado hasta el momento para transitar de arenales a bosques incluyen: 1) herbivoría por mamíferos pequeños que pueden eliminar completamente las plantaciones de algunas especies (*e.g.* conejos han eliminado plantaciones de *Lupinus elegans*), 2) la profundidad de la capa de arena, que representa una barrera no lineal para el establecimiento de las especies vegetales (Gómez Romero et al. 2006; Alejandre Melena 2004); 3) las bajas concentraciones de nutrientes en la ceniza volcánica y 4) las altas temperaturas que alcanza la arena en la época seca del año, lo cual aparentemente es la barrera más importante (Lindig-Cisneros *et al.,* 2007). Estas barreras sugieren que los arenales constituyen un estado estable pues la composición florística es diferente a la de otros estados, (Galindo-Vallejo *et al*., 2006) además de que se presentan cambios temporales no lineales causados por eventos catastróficos (Scheffer *et al*., 2001). Por tal razón, existen múltiples barreras que impiden que el sistema transite al estado que presentaba antes del cambio y se requiere de intervención para llevarlo a otros estados.

The Ecological Restoration Spectrum

Degree of effort → Degree of degredation ↓	Single action (H_i, S_i, V_i, or F_i)	Multiple actions for 1 component (e.g., H_{ijk})	Single action, >1 component (e.g., $H_i + S_i$)	Multiple actions, >1 component (e.g., $S_{ijk} + V_{ijk}$)	Many actions, all components ($H_{ijk} + S_{ijk} + V_{ijk} + F_{ijk}$)
Major large area intensive damage					
Major–Moderate		B. Wetland and channels excavated from fill, connected to tidal source (H_{ij}) to attract fish		C. Marsh excavated from fill, connected to tidal source (H_{ij}), fertilized (S_i), planted and weeded (V_{ij})	
Moderate					
Moderate–Minor	A. Endangered plant reintroduced (V_i) to existing wetland				
Minor small area minor damage					

Figura 1. Modelo original del espectro de restauración ecológica (Zedler 1999).

Grado de esfuerzo/ grado de perturbación	Una acción	Varias acciones sobre un componente	Una acción en más de un componente	Varias acciones sobre varios componentes	Varias acciones, todos los componentes
Mayor				CUTZATO Protección contra herbivoría (ganado, roedores) y erosión pluvial, reforestación, uso de leguminosas y acolchados, **CF, CB**	
Mayor- moderado	PARIO; Reforestación con árboles resistentes a heladas, **CF**				
Moderado			SAN NICOLAS Sembrado de leguminosa s **CBG, D**	SAN NICOLAS Reforestación, protección contra herbivoría (ganado y tuza), sembrado de leguminosas, **CF, CBG, D**	
Moderado-menor					
menor	BOSQUE Reforestación tradicional en predios con manejo forestal, **CF**				

Figura 2. Modelo de espectro de restauración adaptado para los sitios de Nuevo San Juan Parangaricutiro

CF; cobertura forestal, CBG; ciclos biogeoquímicos, D; diversidad,
SAN NICOLAS; campo agrícola abandonado
PARIO; arenal originado por la erupción del Volcán Paricutín removido en 2000-2004 con fines de construcción de casas
CUTZATO; arenal intacto, remanente de la erupción del Volcán Paricutín.

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